Biosecurity under uncertainty: the influence of information availability and quality on expert decision-making for risk outcomes

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Submitted in fulfilment

of the requirements for the Degree of Doctor of Philosophy

University of Tasmania.

Launceston, Tasmania

March 2012

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STATEMENT OF CO-AUTHORSHIP

The following people and institutions contributed to the publication of the work undertaken as part of this thesis:

Chapter 2:

Dahlstrom, A., C. L. Hewitt, and M. L. Campbell. 2011. A review of international, regional and national biosecurity risk assessment frameworks. Marine Policy 35:208-217.

Dahlstrom, A (60%), Hewitt, CL (20%), Campbell, ML (20%)

Chapter 3:

Dahlstrom, A., C. L. Hewitt, and M. L. Campbell. 2011. Mitigating uncertainty using alternative information sources and expert judgment in aquatic nonindigenous species consequence assessment. Aquatic Invasions (in review).

Dahlstrom, A (60%), Hewitt, CL (20%), Campbell, ML (20%)

Chapter 4:

Dahlstrom, A., C. L. Hewitt, and M. L. Campbell. 2011. The role of uncertainty and subjective influences on consequence assessment by aquatic biosecurity experts. Biological Conservation (in review).

Dahlstrom, A (60%), Hewitt, CL (20%), Campbell, ML (20%)

- CL Hewitt and ML Campbell both contributed to the idea, its formalization and development, and assisted with refinement and presentation.

Chapter 5:

Dahlstrom, A. and C. L. Hewitt. 2011. Evidence of impact: low statistical power leads to false certainty of no impact for nonindigenous species. Nature (in prep).

Dahlstrom, A (80%), Hewitt, CL (20%)

- CL Hewitt contributed to the idea, its formalization and development, and assisted with refinement and presentation.

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ABSTRACT

Alongside climate change and habitat loss, aquatic nonindigenous species (ANS) introductions comprise a large and increasing contribution of the anthropogenic threat to environmental, economic, sociocultural and human health values worldwide. Biosecurity agencies aim to prevent and manage introductions using various tools, including risk assessment. Risk assessment can prioritize threats, but is frequently compromised by uncertainty, often due to information availability, quality and interpretation. Many risk assessment processes lack consistent and transparent treatment of uncertainty, particularly when biosecurity objectives warrant a precautionary approach.

This thesis aims to identify methods for managing uncertainty via an initial review of 14 existing national, regional and international biosecurity instruments. Results from this review found over half of the instruments explicitly included or mentioned precaution, and many instruments acknowledged the potential influence of subjective risk perceptions. Based on these outcomes, this thesis aims to: determine sources of uncertainty; understand the cognitive process of estimating consequence, and therefore risk under uncertainty; and provide transparent methods to reduce uncertainty that allow for precaution, using input from ANS experts in scientific and management fields. Finally, this thesis aims to examine how the frequentist statistical focus on low acceptable rates of Type I errors, most frequently applied in ANS impact research, influences findings of significant impact and the implications for management decisions.

Results of this thesis indicate that the scarcity of ANS impact information constitutes a primary source of uncertainty. When faced with knowledge gaps and other forms of uncertainty, experts tended to assume and assign lower consequence via a ‘hindsight approach’ (assume no impact without sufficient information), which stands opposite to precaution. To mitigate the effects of uncertainty, experts supported the use of alternative information sources, including non-empirical evidence. In practice, the provision of information and group discussion generally increased the consequence estimate, thus suggesting methods that allow functional and, if desired, precautionary consequence assessments despite high uncertainty. In situations of expected ‘low’ certainty, when information is available, my research indicated that an extremely high proportion of statistical analyses of impact had insufficient power to detect an impact, leading to ‘false certainty’ of no impact. This bias toward ‘missing’ impacts, again opposite to precaution, may further prevent appropriate management action.

The thesis concludes with a proposed framework that provides guidance for biosecurity-related research and management using an acceptable level of risk. It provides a transparent process and usable risk outcomes that: (1) integrate scientific process and management objectives; (2) are accountable for and unimpeded by uncertainty; (3) consider the assumptions used by the experts making the assessment; (4) can be adapted according to varying strengths of precaution desired by management; (5) follows World Trade Organization Sanitary and Phytosanitary (SPS) Agreement mandates; and (6) are feasible given time and budget constraints.

ACKNOWLEDGEMENTS

I wish to sincerely thank my supervisor Prof. Chad Hewitt and co-supervisor Prof. Marnie Campbell. I began my PhD with a project that promptly changed within a few weeks of my arrival. They provided me with the freedom to develop a new project, yet provided patient and invaluable guidance along the way that allowed me to arrive at its successful conclusion with some novel insights and appreciation for some of the social underpinnings of science. Their time and effort as mentors and, eventually, colleagues is most gratefully appreciated.

I am grateful to Dr. Greg Ruiz for providing me space at the Smithsonian Environmental Research Center (SERC) and also valuable input into my project during the first year of its development. A. Whitman Miller and Paul Fofonoff, also at SERC, provided useful feedback and collaboration.

I am grateful to the Australian Maritime College, a specialist institute of the University of Tasmania, for financial support via the John Bicknell Research Scholarship, which permitted my study in Australia. I also acknowledge the other sources of funding including the Australian Geographic Society and the University of Tasmania Graduate Research Candidate Conference Fund Scheme. I also thank the Human Research Ethics Committee (Tasmania) Network for ethics approval.

I wish to thank Jan Daniels and Mandy Norton for their kind assistance with all things administrative. I also wish to thank those individuals within each conference that facilitated organization of each workshop: Robyn Draheim (MBIC), Narelle Hall (AMSA), Lisa Carmody and Elizabeth Muckle-Jeffs (ICAIS) and Sonia Gorgula (NIMPCG). I owe the outcomes of Chapters 3 and 4 to all the participants who generously gave their time participating in the surveys and workshops, tirelessly putting up blue and red dots.

To my parents, my endless thanks. It is only as I grow older that I realize the sacrifices you made as individuals and as a couple to get me to where I am. I am sorry I ended up so far away, but thank you for letting me go. To my brothers, your humour and camaraderie have a special place in my heart – I look forward to larger doses in the future. To my friends back in ‘merica, thank you for not forgetting me, down here at the end of the world, and always welcoming me back with big smiles and open arms when I manage a whirlwind visit (and a special thanks to those who ventured down here for a visit, long or short!) To my Tassie friends, thank you for welcoming me to the island and taming my obnoxious accent; you made these three years a joy. And to Tassie – I fell in love with your beautiful wildness like few other places I’ve been; they gave me peace and comfort when nothing else could. And to my partner in all things, Ben (GGB), thank you for your unfailing patience, love and ability to remind me what matters most in life.

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GLOSSARY OF TERMS RELEVANT TO THIS RESEARCH

Term	Definition
α (alpha)	The acceptable rate of Type I errors, or incorrectly rejecting the null hypothesis (Quinn and Keough 2002).
Alien species	Species that spread beyond their native range, not necessarily harmful, or species introduced to a new range that establish themselves and spread; similar terms include exotic species, foreign species, introduced species, non indigenous species, and non native species (Occhipinti-Ambrogi and Galil 2004, Jeschke and Strayer 2005).
ALOP (ALOR)	Acceptable Level of Protection (Acceptable Level of Risk). The level of protection deemed appropriate by the Member establishing a sanitary or phytosanitary measure to protect human, animal or plant life or health within its territory; also known as the acceptable level of risk (WTO 1995).
Ambiguity	The variability of (legitimate) interpretations based on identical observations or data assessments (Klinke and Renn 2002).
ANS	Aquatic Nonindigenous Species. Nonindigenous species (defined below) in marine, brackish or freshwater habitats.
Aquatic biosecurity	National, regional and international efforts to prevent, reduce, and manage the introduction of pests, diseases or unwanted organisms via entry and border surveillance, short-term response and long-term control of established pests.
Nuisance species	See *invasive species*.
Attitude	Evaluative reaction(s) to an object or behaviour that is based on beliefs about that object or behaviour and which is associated with behaviour toward the attitude object (Clayton and Myers 2009).
Availability heuristic	The tendency for events and outcomes to appear more probable when they come to mind more easily (Clayton and Myers 2009).
β (beta)	The acceptable rate of Type II errors, or incorrectly accepting a false null hypothesis (Quinn and Keough 2002).
Ballast water	The uptake and release of organisms during ballasting and de-ballasting operations (respectively), which are necessary to maintain trim, stability, propeller immersion, and safe levels of hull stress during travel or in port (Victorian Government 2006).
Biological control	The release of one species to control another (Carlton 2001).
Biological diversity or biodiversity	Used to describe species richness, ecosystem complexity, and genetic variation (Allaby 1998).

Glossary cont.

Term	Definition
Biological invasion or bioinvasion	A broad term that refers to both human-assisted introductions and natural range expansions (Carlton 2001).
Categorical descriptors	Categorical definitions of impact defined in a qualitative (e.g., low, medium or high) or quantitative manner. In semi-quantitative assessments, the definitions of consequence categories are often based on “threshold values”, often in a combination of qualitative and numerical terms. Threshold values often include measures of magnitude, spatial extent of the impact (e.g. local, regional, or global), temporal scale of the impact (e.g. temporary or permanent), and resilience of the system to impact (e.g. the potential of the effected entity to recover). Each threshold description may contain several conditions, only one of which must be met in order to classify the impact to that category (Campbell 2005, 2008, Campbell and Gallagher 2007).
CBD	Convention on Biological Diversity. An international treaty to sustain the diversity of life on Earth.
Clean lists	A list of permitted species for introduction or import based on invasion history or characteristics (Ruesink et al. 1995, Simberloff et al. 2005); assumes guilty until proven innocent.
Cognitive bias	A patter of judgment that occurs when people rely on a limited number of heuristic principles that reduce the complex tasks of assessing probabilities and predicting values to simpler judgmental operations (Tversky and Kahneman 1974).
Community	Any grouping of populations of different organisms that live together in a particular environment (Allaby 1998).
Consequence (assessment)	The assessment (and related terms, e.g., ‘core values’, ‘subcomponents’ and ‘categories’) of potential impacts posed by a threat and which is combined with likelihood assessment to produce a risk estimate (Campbell 2005, 2008, Campbell and Gallagher 2007, Campbell and Hewitt 2008).
Consequence tables	The combination of categorical descriptions of consequence and associated threshold descriptions. The consequences of a species for each value area are then combined to give an idea of that species’ overall consequence to a region (Campbell 2005, 2008, Campbell and Gallagher 2007, Campbell and Hewitt 2008).
Core value bias	Any difference in perceived consequence due to different valuation of the area of impact.

Glossary cont.

Term	Definition
Core values	The main value types against which impacts are assessed (e.g., Campbell 2008). They can include environmental, economic, social, cultural and human health. Environmental impacts of ANS can be ecological (abundance and distribution of organisms), biological (the organisms themselves), chemical (processes such as bioaccumulation of toxins) or physical (processes such as disturbance); qualitative or quantitative; structural or functional (Ward 1978). Economic impacts are effects on humans which alter their activities in ways that affect their incomes and expenditures of money (Fofonoff et al. 2003). Social impacts affect the values placed on a location or species in relation to human use for pleasure, aesthetic, and generational values (Campbell 2008). Cultural impacts affect aspects of the aquatic environment that represent an iconic or spiritual value, including those that create a sense of local, regional, or national identity (Campbell 2008). Human health impacts affect the value of a safe and healthy society shared equally across generations and socio-economic groups (Hewitt et al. 2010).
Cryptogenic species	Species that are neither clearly indigenous or nonindigenous (Carlton 1996a)
Decision theory	A multidisciplinary set of theories that describe the use of various principles in choosing one of multiple available options based on the perceived state of nature and potential consequences, often in an effort to maximize utility or rationality (Chernoff and Moses 1959).
Delphic process (modified)	A process used to make decisions and predictions in conditions of scarce and/or highly uncertain information inappropriate for traditional scientific methods, which uses expert revision of judgment based on input and opinion of other experts to reach consensus where possible and identify areas of disagreement where consensus is not possible, with a subsequent reduction in overall uncertainty (Webler et al. 1991). While the original process used anonymous expert input, a modified Delphic process uses expert input via group workshop process (Webler et al. 1991).
Dirty lists	A list of prohibited species for introduction or import based on invasion history or characteristics (Ruesink et al. 1995, Simberloff et al. 2005); assumes innocent until proven guilty.
DMURI	Decision Making Under Risk and Ignorance.
Ecological risk assessment	An evaluation of the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (US EPA 1992).

Glossary cont.

Term	Definition
Economic valuation	Attempts to assign quantitative values to the goods and services provided by environmental resources, whether or not market prices are available. The economic value of any goods or services is generally measured in terms of what resource users or society at large are willing to pay for the commodity, minus what it costs to supply (Bonzon and Cochrane 2003).
Ecosystem	A discrete unit, or community of organisms and their physical environment (living and non-living parts), that interact to form a stable system (Allaby 1998).
Effect size	A statistical measurement of the difference between two populations that provides an estimate of the magnitude and direction of an effect (Nakagawa and Cuthill 2007).
Endpoints	The values affected by the hazard, which a risk assessment aims to protect and by which impacts are measured (US EPA Risk Assessment Forum 2003).
Environmental matching assessment	A risk assessment approach that compares environmental conditions including temperature and salinity between donor and recipient regions. The degree of similarity between the locations provides an indication of the likelihood of survival and the establishment of any species transferred between those locations (Herborg et al. 2007).
Epistemic uncertainty	Epistemic uncertainty stems from a lack of knowledge and can be ameliorated via additional research or similar efforts (Walker et al. 2003).
Expected utility	The expected utility of an act can be calculate once the probabilities and utilities of its possible outcomes are known, by multiplying the probability and utility of each outcome and then summing all terms into a single number representing the average utility of the act (Peterson 2009).
Hazard	An object or event that has the potential to cause harm under specific conditions that allow that risk to be realized; in order to assess the hazard, both the object (e.g. a vector, trade route, or species) and the conditions (e.g. the recipient port environment) are considered (Hewitt and Hayes 2002).
Heuristics	Heuristics are learned, declarative or procedural knowledge structures stored in memory (e.g., "rules of thumb", judgmental shortcuts, biases, educated guesses, intuitive judgments or simply common sense tools) that have been learned and internalized by the individual (e.g., ‘length implies strength’) to deal with an increasingly complex world, in which individuals are forced to make decisions using either an overwhelming or insufficient amount of information (Chaiken et al. 1989, Chen and Chaiken 1999).
Hindsight approach	In an ANS impact assessment context, when information is lacking or scarce, the assumption that a species is “innocent until proven guilty”.

Glossary cont.

Term	Definition
HSM	The Heuristic-Systematic Model attempts to explain how individuals make decisions under risk and ignorance (Trumbo 1999). The HSM identifies two methods by which people make judgments: systematic processing (a comprehensive analysis) and heuristic processing (a shortcut-based analysis; this occurs if an individual is unwilling or unable to take the time or make the effort to carefully consider the evidence) (Chen and Chaiken 1999, Trumbo 2002).
Hypothesis testing	see *NHST*
Impact assessment	Impact refers to the assessment of impacts to environmental, economic, social, human health, or cultural values caused by ANS, which contributes to the formal consequence assessment.
Import Health Standards	Legislative procedural documents established to ensure that the internationally agreed standard for quarantine and scientific evaluation are met, in order to reduce the unwarranted restrictions of trade when importing goods (Campbell 2009).
Incursion	See *Introduction*
Indigenous	A species that occurs naturally in an area; also known as native (Allaby 1998).
Information type/source	In this context, possible sources of knowledge for use in a consequence or risk assessment (e.g., harbour manager observations, grey literature, and experimental research).
Intentional introduction	A species that is brought to a new area, country, or bioregion for a specific purpose, such as for a garden or lawn; a crop species; a landscaping species; a species that provides food; a groundcover species; for soil stabilization or hydrological control; for aesthetics or familiarity of the species; or other purposeful reasons (Booth et al. 2003).
Introduced species	This terms means those species that have been transported by human activities, either intentionally or unintentionally, into a region in which they did not occur in historical time and are now reproducing in the wild (Carlton 2001). Similar terms include alien, exotic, foreign, nonindigenous, and non-native.
Introduction	The human mediated movement of an animal to an area outside its natural range (Hewitt et al. 2010).
Invasion	The expansion of a species into an area not previously occupied by it (Booth et al. 2003).

Glossary cont.

Term	Definition
Invasive species	Generally, this term refers to a subset of native or non-native plants or animals that are cause economic or environmental harm or harm to human health (Executive Presidential Order 1999). Commonly, widespread nonindigenous species that have adverse effects on the invaded habitat (Colautti and MacIsaac 2004). Similar terms include pest and nuisance.
Likelihood	In an aquatic biosecurity context, the probability of ANS incursion or establishment, described in qualitative, semi-quantitative or quantitative terms.
Maximin principle	A decision rule sometimes used in decisions under ignorance, which holds that one should maximise the minimal value obtainable in each decision. Hence, if the worst possible outcome of one alternative is better than that of another, then the former should be chosen (Peterson 2009).
National sovereignty	In this context, the WTO-provided right of any government to set the level of health protection it deems appropriate, but to ensure that these sovereign rights are not misused for protectionist purposes and do not result in unnecessary barriers to international trade (WTO 1998).
Native species	See *indigenous species*
NEMESIS	National Exotic Marine and Estuarine Species Information System; a database developed by the Smithsonian Environmental Research Center (SERC), http://invasions.si.edu/nemesis/
NHST	Null Hypothesis Significance Testing. The statistical evaluation of whether a set of results differs from a pre-identified null hypothesis, based on the probability (*p*-value) that the findings are unlikely to be within the population of the control (Nakagawa and Cuthill 2007).
NIMPIS	National Introduced Marine Pest Information System; a database developed by the Australian Government, http://adl.brs.gov.au/marinepests/
Non-native species	See *nonindigenous species*
Nonindigenous species	Species that have been transported by human activities – intentionally or unintentionally – into a region in which they did not occur in historical time (Hewitt et al. 2010). Similar terms include alien, exotic, and foreign.
Norms	Customary rules of behaviour that coordinate our interactions with others (Lewis 1969).
Ontological uncertainty	Stems from the inability to fully describe a variable and complex environment and cannot be eliminated (Walker et al. 2003).
Pathway	The vector, purpose (the reason why a species is moved), and route (the geographic corridor from one point to another) (Carlton 2001).

Glossary cont.

Term	Definition
Pest (IPPC)	Any species, strain or biotype of plant, animal or pathogenic agent injurious to plants or plant products (FAO 1997).
Population	A group of potentially inter-breeding individuals of the same species found in the same place at the same time (Booth et al. 2003).
Power analysis	A statistical procedure that uses sample size (n), significance criterion (α and β), effect size (ES) and σ (population standard deviation) to determine power (di Stefano 2003, Nakagawa and Cuthill 2007).
Power	The probability of correctly rejecting the hull hypothesis and the complement of the Type II error rate β, 1-β (Lehmann and Romano 2005, Nakagawa and Cuthill 2007).
Precaution	The stance that, "Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (United Nations General Assembly 1992). It often includes reversing the burden of proof, i.e., proving that an activity down not cause harm before proceeding. Often described in terms of the *precautionary principle* or *precautionary approach*.
Quarantine	Official confinement of regulated articles for observation and research or for further inspection, testing and/or treatment (IPPC Secretariat 2007).
Recognition heuristic	A "fast and frugal" heuristic by which individuals rank an object as more fitting of a criterion based on their recognition of it (e.g., ranking a city as greater in population based on recognizing its name) (Goldstein and Gigerenzer 2002).
Risk (decision theory)	A decision under risk occurs when the decision maker knows the probability and utility of all possible outcomes (Peterson 2009).
Risk (biosecurity)	The possibility that human actions or events lead to consequences that harm aspects of things that human beings value (Klinke and Renn 2002).
Risk analysis	A process comprised of risk assessment, risk management, risk communication, and risk policy (Byrd and Cothern 2000).
Risk assessment	The process of determining the probability (likelihood) and impacts (consequences) of that event (Hayes 1997).
Risk communication	A process that helps clearly express the risk to the relevant stakeholders and/or public (Byrd and Cothern 2000).
Risk management	A process that involves analysing and choosing the best options to reduce, eliminate or otherwise address the risk (Byrd and Cothern 2000).

Glossary cont.

Term	Definition
Risk matrix	Tables with vertical and horizontal headings that correspond to likelihood (aka probability, frequency, etc) and consequence (aka impact, severity, etc) in order to provide a risk estimate (e.g., Campbell 2008, Standards Australia 2009).
Risk perception	The multidisciplinary study of how and why people perceive risks, in recognition of the fact that this process occurs differently depending on the nature of the risk and the individual (Cohrssen and Covello 1989).
Risk policy	Risk policy surrounds the entire process of risk analysis, e.g., in developing guidelines for each component to improve the structure and process of risk analysis (Andersen et al. 2004).
Satisficing	A term from “satisfy” and “suffice”, in which respondents choose not to expend energy making the optimal decision, instead merely making a choice that seems adequate (Krosnick 1999).
SIF	Subjective Influencing Factors. The values, attitudes, norms and biases that impact information processing and decision making within individuals and agencies responsible for estimating and managing risks.
Significance testing	see *NHST*
Species name bias	A difference in assessed consequence based on the species or genus name of the ANS.
Species	A group of organisms formally recognized as distinct from other groups; the basic unit of biological classification, defined by the reproductive isolation of the group from all other groups of organisms (Allaby 1998).
Species-specific assessment	A risk assessment approach that uses information on life history and physiological tolerances to define a species’ physiological limits and thereby estimate its potential to survive or complete its life cycle in the recipient environment. That is, a comparison of individual species characteristics with the environmental conditions in the recipient port, to determine the likelihood of transfer and survival (IMO 2007).
SPS Agreement	Agreement on the Application of Sanitary and Phytosanitary Measures. A WTO framework that sets out the basic rules for food safety and animal and plant health standards, including measures taken to protect the health of fish and wild fauna, as well as of forests and wild flora. Sanitary and phytosanitary measures are defined as any measures applied: to protect human or animal life from risks arising from additives, contaminants, toxins or disease-causing organisms in their food; to protect human life from plant- or animal-carried diseases; to protect animal or plant life from pests, diseases, or disease-causing organisms; and to prevent or limit other damage to a country from the entry, establishment or spread of pests (WTO 1998).

Glossary cont.

Term	Definition
Subcomponents	A specific type of impact, within a core value, that has a unique unit, method and description for the measurement of consequence or impact.
Systematic measurement error	Error that results from biases or imperfections in collecting or interpreting measurements (Klinke and Renn 2002).
TPB	The Theory of Planned Behaviour is a modification of TRA that adds perceived behavioural control as an additional construct to determine the behavioural intention (Ajzen 1991).
TRA	The Theory of Reasoned Action posits that the attitudes and subjective norms surrounding an action combine to produce the behavioural intention (i.e., decision to perform an action) (Ajzen 1991).
Type I error	In an impact assessment context, incorrectly assigning an impact to a species.
Type II error	In an impact assessment context, incorrectly assigning no impact to a species.
Uncertainty (decision theory)	A decision under uncertainty occurs when the decision maker knows the utility of all possible outcomes, but not their probabilities (Peterson 2009).
Uncertainty	Any deviation from the unachievable ideal of completely deterministic knowledge of the relevant system (Walker et al. 2003).
Unintentional introduction	An introduction of nonindigenous species that occurs as the result of activities other than the purposeful or intentional introduction of the species involved, such as the transport of nonindigenous species in ballast or in water used to transport fish, molluscs or crustaceans for aquaculture or other purposes (US EPA 2000).
Utility (decision theory)	The more an object is desired, the higher is its utility. Utility is measured on some utility scale, which is either ordinal or cardinal (Peterson 2009).
Values	General, stable, strongly held judgments or preferences for end states or ways of acting that serve as goals that apply across different contexts (Clayton and Myers 2009).
Vector	The physical means or agent by which a species is transported, such as ballast water, ships' hulls, boats, hiking boats, cars, vehicles, packing material, or soil in nursery stock (Carlton 2001).

Glossary cont.

Term	Definition
Vessel fouling	The association of aquatic organisms with objects immersed in salt water, including the hulls and ancillary gear of commercial and other vessels (AMOG Consulting 2001). Fouling species (including small fish, barnacles, mussels, sponges, algae, crabs, and sea squirts) foul ships via attaching to the wetted surface areas, or finding refuge within the matrix of the fouling community and protected nooks and crannies (e.g., sea chests) (AMOG Consulting 2001, Coutts et al. 2003).
WTO	World Trade Organization. An international organization dealing with the rules of trade between nations.

CHAPTER 1. GENERAL INTRODUCTION

Aquatic nonindigenous species

Within the suite of current environmental issues, aquatic nonindigenous species (ANS) remain one of the primary threats (Lubchenco et al. 1991, Bax et al. 2003). While climate change is the oft-cited driver of global change, nonindigenous species pose a serious threat in their own right (e.g., Ruiz et al. 1997, Carlton 2001, Hewitt et al. 2004a), as well as synergistically interacting with and augmenting the effects of climate change and other threats to biodiversity such as land use change, climate change, overexploitation, and pollution (Halpern et al. 2007). Damages imposed by introduced species will certainly effect the more 'ecocentric' values (such as biodiversity), but with impacts extending to human well-being – including food security, basic material for a good life, health, good social relations, and freedom of choice and action – they will threaten the very way we live (Millennium Ecosystem Assessment 2005).

While research and management initiatives to control and prevent aquatic invasions have only occurred for a few decades, human-mediated species introductions have occurred for thousands of years, with introduction rates accelerating during the last 200 years (Crosby 1986). The increasing rates of introductions are due to a variety of factors, including stressed ecosystems (Occhipinti-Ambrogi and Savini 2003) and increasing quantity and quality of vector-based transfer (current figures estimate over 10,000 species in transit at any given time) (Carlton 1999, Hulme 2009). Despite the thousands of documented ANS, these figures may underestimate actual numbers given uncertain systematics, confounding biogeography and insufficient sampling (Carlton 1996a, Ruiz and Hewiit 2002, Hewitt et al. 2004a).

Regardless of the potential for under-documentation of the species, the threat remains global; ANS have impacted every marine bioregion in the world (sensu the IUCN WCPA – Marine Plan of Action adapted from Kelleher et al. 1995) (Laffoley 2006; Figure 1.1). As a region with species arriving only recently, impacts to the Antarctic (IUCN Bioregion 1) by ANS such as *Hyas araneus* remain unknown (Tavares and Melo 2004). *Enteromorpha prolifera* degrades sandflat communities in the near-Arctic (IUCN Bioregion 2) (Bolam et al. 2000); *Hydroides dianthus* fouls infrastructure in the Mediterranean (IUCN Bioregion 3) (Galil 2000); *Hemigrapsus sanguineus* alters structure of rocky intertidal communities in the northwestern Atlantic Ocean (IUCN Bioregion 4) (Gerard et al. 1999); *Polysiphonia harveyi* fouls ropes and pontoons in northeast Atlantic marinas (IUCN Bioregion 5) (Maggs and Stegenga 1998); *Acartia tonsa* dominates communities in the Baltic Sea (IUCN Bioregion 6) (Leppakoski et al. 2002); *Perna viridis* displaces native species in the Caribbean (IUCN Bioregion 7) (Buddo et al. 2003); *Caulerpa racemosa* var. *cylindracea* spreads over substrate in the Canary Islands (IUCN Bioregion 8) (Verlaque et al. 2004); *Limnoperna fortunei* impedes water-treatment plants, industrial refrigeration systems and power stations in the South Atlantic (IUCN Bioregion 9) (Darrigran 2002); *Mytilopsis sallei* forms thick fouling communities on vessels in India (IUCN Bioregion 10) (Morton 1981); *Sparus aurata* may impact fisheries in the Arabian Sea (IUCN Bioregion 11) (Global Invasive Species Database, http://www.issg.org/database); *Salvinia molesta* degrades aquaculture and tourism in Lake Naivasha, Kenya (IUCN Bioregion 12) (Caspers 1976); *Oreochromis* spp. has contributed to the decline or extinction of several species in the Philippines (IUCN Bioregion 13) (Pullin et al. 1997); *Chthamalus proteus* forms almost 100% cover in some Hawaiian intertidal zones (IUCN Bioregion 14) (Zabin and Altieri 2007); *Corbula amurensis* has disrupted trophic interactions in the San Francisco Bay (IUCN Bioregion 15) (Alpine and Cloern 1992); *Pyromaia tuberculata* dominates Tokyo Bay, Japan (IUCN Bioregion 16) as the most abundant crab (Wahitani 2004); *Codium fragile* invasion threatens the persistence of *Gracilaria chilensis* farms in northern Chile (IUCN Bioregion 17) (Neill et al. 2006); and *Asterias amurensis* predates on commercially farmed bivalves in Tasmania (IUCN Bioregion 18) (Ross et al. 2002).

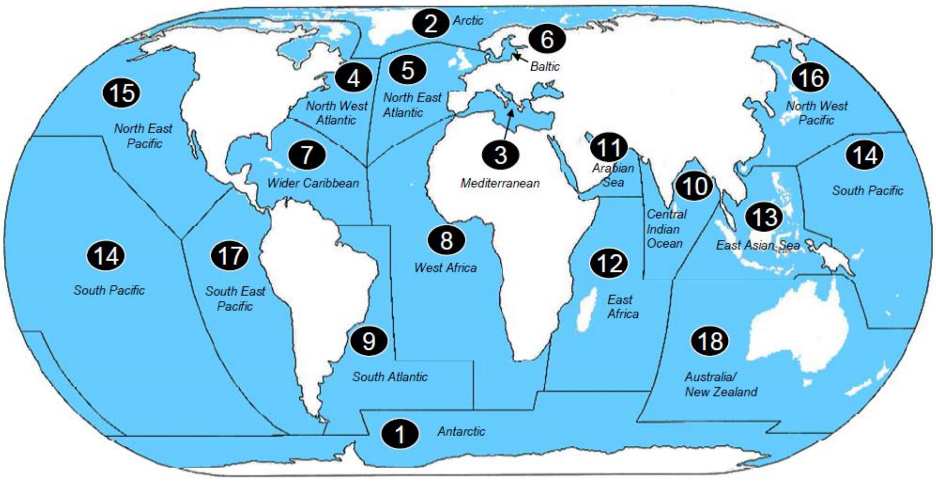

Figure 1.1. The global marine bioregions derived from an ecosystem-based approach (used by IUCN, based on Kelleher et al. 1995).

Within IUCN Bioregion 18, the effects of introduced marine species in all Australian bioprovinces (Figure 1.2) have been well documented (Hewitt et al. 1999, Hewitt 2002, Hayes et al. 2004, Hewitt and Campbell 2008, Neil et al. 2008). For example, *Bugula neritina* heavily fouls ports and harbours in the Solanderian province (Keough and Ross 1999); *Alexandrium minutum* produces paralytic shellfish poisons in the Peronian province (Hallegraeff et al. 1991); *Asterias amurensis* reduces survivorship of recently settled juveniles of the commercial bivalve *Fulvia tenuicostata* in the Flindersian province (Ross et al. 2002); and *Zoobotryon verticillatium* fouls ports and harbours in the Dampierian province (Wells et al. 2009).

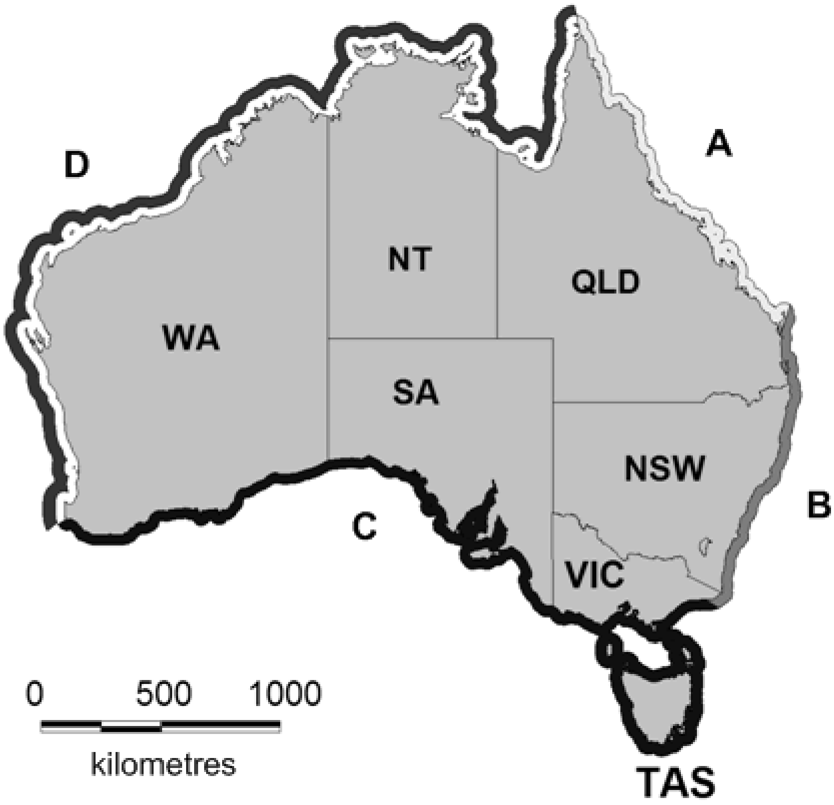

Figure 1.2. The four major biogeographic provinces of Australia: A Solanderian; B Peronian (includes Lord Howe and Norfolk Islands); C Flindersian; and D Dampierian (includes Cocos, Keeling and Thursday Islands and Ashmore Reef) (Campbell and Hewitt 2011, as adjusted from Poore 1995).

With the increasing number of introductions comes an increasing variety and magnitude of potential impacts from ANS. These impacts occur over a variety of values (environmental, economic, social, cultural and human health, hereafter referred to as 'core values'; Campbell 2008) and scales. In an environmental context, ANS can have negative effects at the species level (e.g., *Sargassum muticum* dominates the eelgrass *Zostera* via prevention of recolonization in Brittany, France; den Hartog 1997), community level (e.g., *Musculista senhousia* alters community assemblages via their mat-forming byssal threads in Tamaki Estuary, New Zealand; Creese et al. 1997); and ecosystem level (e.g., *Potamocorbula amurensis* decreases pelagic production within the San Francisco Bay through filtration of the phytoplankton bloom; Alpine and Cloern 1992). As ANS expand their relative ranges, similar suites of ANS are becoming the dominant species at local and regional scales, leading to a loss of community biodiversity and eventual 'biotic homogenization' (McKinney and Lockwood 1999). To make matters worse, ANS impacts can synergistically combine with other human-mediated impacts such as habitat destruction, pollution and climate change (Bianchi and Morri

2000), or facilitate the success of other invaders, increasing the overall impact and potentially leading to 'invasional meltdown' (Simberloff and Von Holle 1999). As eradication, or control of these species is extremely difficult, costly, and oftentimes impossible once established, prevention is not only more economically efficient, but also often necessary to prevent impacts from occurring (Mack et al. 2000). However, accurate prediction as to which species will establish, spread and have impacts in order to aid prevention measures remains generally unsuccessful (Carlton 1996b, Ricciardi 2003, Guo 2006), particularly given the lack of necessary information (Nyberg and Wallentinus 2005).

Despite the difficulties in identifying invasive ANS, measures to prevent these ANS are recognized as necessary given the potential severity of their impacts (Bax et al. 2003). These measures often focus on ANS vectors, as a limited number of vectors are responsible for the majority of unintentional introductions, namely ballast water and biofouling (Rigby et al. 2003, Hewitt and Campbell 2008). Ballast water (as a vector) refers to the uptake and release of organisms during ballasting and de-ballasting operations (respectively), which are necessary to maintain trim, stability, propeller immersion, and safe levels of hull stress during travel or in port (Victorian Government 2006, IMO 2011). Biofouling (i.e., vessel fouling or hull fouling) refers to the association of aquatic organisms with objects immersed in water, including the hulls and ancillary gear of commercial and other vessels, as well as sea chests (Coutts et al. 2003, Hewitt et al. 2004a, Hewitt et al. 2004b, Tavares and Melo 2004, Coutts and Dodgshun 2007). Fouling species and associated species (including small fish, barnacles, mussels, sponges, algae, crabs, and sea squirts) foul ships via attaching to the wetted surface areas, or finding refuge within the matrix of the fouling community and protected areas (e.g., sea chests) (Coutts et al. 2003). Once ANS enter ballast tanks or colonize a vessel, they are carried between ports and inoculation occurs via release (ballast water) or spawning and/or physical removal (of fouling), with subsequent opportunity for establishment, spread and impact (Hewitt et al. 2009).

While ballast water has been often cited as the primary vector for ANS transfer, biofouling has also been identified as a significant vector (Woods Hole Oceanographic Institution 1952, Cohen and Carlton 1995, Cranfield et al. 1998) that in some areas is responsible for more introductions than ballast water (Eno et al. 1997, Coutts 1999, Drake and Lodge 2007, Hewitt and Campbell 2008). For example, biofouling is potentially responsible for 78.3% of ANS in Port Phillip Bay, Australia (Hewitt et al. 2004a); 69% of introduced marine species in New Zealand (Cranfield et al. 1998); and over 50% of vessel-mediated introductions in the North Sea (Gollasch 2002). Biofouling species also have significant impacts on the marine environment, natural resources and industries (e.g. aquaculture, fisheries, tourism, commercial shipping, and marine infrastructure). In Australia, eradication and control of the black striped mussel, *Mytilopsis sallei*, alone resulted in costs over A$2.2 million and death of all other organisms in the treated area (Willan et al. 2000, Bax et al. 2002). Similarly, in the United States, damage from the zebra mussel, *Dreissena polymorpha*, and quagga mussel, *Dreissena rostriformis bugensis,* has exceeded US$1 billion (Pimentel 2005).

In addition to the overall increase in vector strength cited above, the risk of introductions due to biofouling may increase due to several factors. First, the *International Convention on the Control of Harmful Anti-fouling Systems on Ships* (enforcement initiated in 2008) bans the use of a common anti-fouling system (AFS), tributyltin (TBT). The absence of an effective and low-cost alternative AFS is likely to result in an increased level and diversity of fouling and introduction potential (Nehring 2001, Lewis et al. 2003), as hulls without TBT as an anti-foulant have shown to have greater biomass than control panels (Jelic-Mrcelic et al. 2006). Second, the 2007-2009 global economic recession has slowed or halted a significant amount of vessel traffic. During the first quarter of 2009, 10% of containerships and 25% of refrigerated vessels were taken out of service and anchored for months near Malaysia, Indonesia and the Philippines due to low anchoring fees (Floerl and Coutts 2009). This may increase the risk of ANS transfer as the economy improves and these vessels re-enter trading activity, as time spent at anchor allows for an increased accumulation of fouling species and can render AFS less effective (Coutts 1999). This is particularly relevant given that southeast Asia is an important trading partner with Australasia, Europe and the Americas, with a number of dominant shipping hubs (e.g., Singapore, Hong Kong, Tanjung Pelepas; Slack and Wang 2002, Lee et al. 2008). It is exacerbated by the cost and time commitments for dry-docking to properly remove the fouling and re-apply AFS. Many ships may undergo in-water cleaning, which has the potential to trigger a reproduction event or remove viable adult organisms with the potential to establish (Floerl and Coutts 2009).

Management of the ANS threat via aquatic biosecurity

Vectors such as biofouling are commonly managed under a suite of tools collectively known as aquatic biosecurity. Aquatic biosecurity involves national, regional, and international efforts to prevent, reduce and manage the introduction of pests and diseases in order to reduce the threat to core values (Hewitt et al. 2004b). This is done via entry and border surveillance and the provision for short-term response and long-term control of established pests (Hewitt et al. 2004b). Biosecurity responds to both intentional (e.g., bioterrorism) and unintentional (e.g., vessel-mediated ANS) threats (Hewitt et al. 2004b).

Elements of biosecurity include pre-border, border and post-border management (Hewitt et al. 2004b). Pre-border management involves understanding and predicting the relative risks presented by various pathways or species, then using this information to manage and minimize harm. Pre-border tools include vector-based activities such as ballast water exchange and hull cleaning before arrival, as well as tools often utilized by the receiving entity, including Import Health Standards and risk assessment (Hewitt 2003b, Hewitt et al. 2004b). Border management addresses the issue at the stage of arrival and includes treatment of fouling or ballast water, as well as education and outreach efforts to prevent future threats. Finally, post-border management depends on detection, eradication and control activities (Hewitt et al. 2004b). As prevention is often the most technically and economically feasible option, pre-border management, and particularly risk assessment, is often the focus of biosecurity efforts.

Risk assessment is an important tool in aquatic biosecurity for several reasons. Namely, risk assessment facilitates efficient and effective ANS management by allowing managers to determine the relative risk of various species, pathways, and vectors, and thus effectively allocate available resources (Andersen et al. 2004). In addition, risk assessment is required by the World Trade Organization (WTO) Sanitary and Phytosanitary (SPS) Agreement to justify national or regional biosecurity policies that may affect trade (e.g. the development or review of import standards, surveillance programs, and incursion responses) (Campbell et al. 2009). The SPS Agreement allows national sovereignty in setting the acceptable risk, but specifies that these policies (based on the associated risk assessments) must be science-based and transparent, minimize negative effects on trade, and make an attempt to cooperate and harmonize with other international standards (WTO 1995). Given the efficiency of prevention versus control or removal efforts (Bax et al. 2003), the use of risk assessment for aquatic biosecurity is often related to prevention measures such as: the development of Import Health Standards (Campbell 2009); the determination of likely species of concern or 'next pests' (e.g., Hayes and Sliwa 2003); and the assessment of which vectors and pathways present the greatest risk (e.g., GloBallast in Clarke et al. 2004, Hewitt and Campbell 2007).

Risk Assessment Background

Risk is present in many common actions and events and consequently, risk assessment often occurs informally to compare the potential negative and positive trade-offs of a threat (Tulloch and Lupton 2003). A formal risk assessment is an essential element of the decision-making process because it clearly defines the components of the decision involved (Williams et al. 2008). This helps take into account all potential impacts (National Research Council 1996a, Byrd and Cothern 2000), including those on core values that may have gone unrecognized without a formal impact assessment process. Risk assessment is also a valuable tool for managing threats efficiently by allowing managers to determine what risks are most significant, the magnitude of that significance, and using this knowledge to subsequently set management priorities (Byrd and Cothern 2000). Flexibly managing risk in this manner can save time and money, as it allows management to identify and respond to the highest-risk threats (Haugom et al. 2002).

Risk terminology and definitions vary according to differences in the context or field, the risk assessor's preferences, and the view of risk as a function of probability (the likeliness that an event will occur) or utility (a combination of the likelihood and impacts of that event) (Shrader-Frechette 1991, Byrd and Cothern 2000). These definitions include "the possibility of loss or injury" (Merriam-Webster 2011); "the probability of future loss" (Byrd and Cothern 2000); "effect of uncertainty on objectives" (Standards Australia 2009); and "a concept used to give meaning to things, forces, or circumstances that pose danger to people or to what they value" (National Research Council 1996a). For consistency, risk is defined within this thesis as the possibility that human actions or events lead to consequences that harm aspects of things that humans value (*sensu* Klinke and Renn 2002). Risk assessment is defined as the process of determining the probability (likelihood) and impacts (consequences) of that event (Hayes 1997).

Risk assessment is part of the risk analysis process, which is comprised of risk assessment, risk management, risk communication, and risk policy[1] (Byrd and Cothern 2000, Arthur et al. 2009). Risk management involves analysing and choosing the best options to reduce, eliminate or otherwise address the risk. This involves weighing a variety of options (including take no action, gather more information, or find methods to reduce the risk) and implementing the most effective option (Byrd and Cothern 2000, Andersen et al. 2004). Risk communication helps to clearly express the information and consequences surrounding the risk to the relevant stakeholders and/or public (Morgan et al. 2002). Risk policy surrounds the entire process of risk analysis and includes developing guidelines for each component to improve the structure and process of risk analysis (Andersen et al. 2004).

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[1] There is some debate on the respective use of "risk analysis" and "risk assessment". Most risk analysts use the definitions provided in Byrd and Cothern (2000), but some organizations (such as the U.S. Department of Defence) reverse the definitions so that risk assessment refers to the entire process (i.e. risk analysis, risk management, risk communication and risk policy). For the purposes of this thesis, "risk analysis" refers to the entire process and "risk assessment" refers to the process of determining the likelihood and consequence.

Types of risk assessments

Because of the wide variety of factors influencing risk assessment (e.g., discipline containing the risk, cultural values and desired outcomes) there is no standardized method or framework. However, certain cross-cutting concepts do exist; a comprehensive risk assessment process generally includes the following components: (1) identifying endpoints; (2) identifying hazards; (3) determining likelihood; (4) determining consequences; and (5) calculating risk (Standards Australia 2009).

1) Identifying endpoints. Assessment endpoints are the values (defined via a specific entity and its measurable attributes) potentially affected by a hazard that the risk assessment aims to protect (Sergeant 2002). For example, a biosecurity risk assessment may measure the risk of an ANS by its effects on biodiversity or water quality. The endpoints should be ecologically and managerially relevant, as well as susceptible to the hazard (Sergeant 2002). The choice of endpoints is a result of the assessor's values and subjective judgment; though each field may have suggested endpoints, there is generally no standardized list. Because of this subjectivity and dependence on political, social and other considerations, the endpoint(s) and acceptable levels of impact to the endpoint(s) should be externally established before the risk assessment is underway (Hayes 1997). In ecological and aquatic biosecurity risk assessment, this is a challenging process because the endpoints are often diverse, numerous and may include subcomponents[2] from each of the core values[3] (e.g., biodiversity within the environmental core value). However, establishing endpoints is a useful and necessary step to meet legal requirements, set limits for damage, serve as models for the creation of additional, situation-specific endpoints, further develop risk assessment frameworks, facilitate action by risk managers and set standards for monitoring (Suter 2000).

2) Identifying hazards. A hazard consists of an object or event that has the potential to cause harm under specific conditions that allow that risk to be realized. In order to assess the hazard, both the object (e.g. a vector, trade route, or species) and the conditions (e.g. the recipient port environment) are considered (Hewitt and Hayes 2002).

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[2] A specific type of impact, within a core value, that has a unique unit, method and description for the measurement of consequence or impact (Campbell 2005, Campbell and Gallagher 2007).

[3] The main types of values (i.e., things that are important to people, government, industry etc) that impacts are assessed against. These include environment, economic, social, cultural and human health factors (e.g., Campbell 2008).

3) Determining likelihood. Once the hazards are identified, risk assessment requires an estimation of an event's likelihood (e.g., the probability of ANS incursion or establishment). The likelihood is usually described in qualitative, semi-quantitative, or quantitative terms.

4) Determining consequences. The consequences are the impacts or effects of the hazard on a range of values (as defined by the assessment endpoints). The consequence assessment can include descriptions of the impact magnitude, frequency, spatial extent, and duration/reversibility. While generally negative, it should be established beforehand if the assessment is to include positive and negative, or negative only impacts. Consequence assessment requires understanding of the baseline conditions, analysis of the actual impacts, and a determination of the significance of the consequences (Westman 1985).

5) Calculating risk. For each core value, the likelihood estimates are considered against the consequence estimates to produce a final risk estimate. This is often done quantitatively or qualitatively using risk matrices. Risk matrices are tables with vertical and horizontal headings that correspond to likelihood (i.e., probability, frequency, etc) and consequence (i.e., impact, severity, etc) (e.g., Campbell 2008, Standards Australia 2009). Each cell of the table corresponds to a combination of these two factors (often coloured green, yellow and red) and provides a risk estimate that can be used to determine appropriate management actions (e.g., Table 1.1) (Cox 2008).

Table 1.1. Qualitative risk matrix (Standards Australia 1999)

	Consequences				
Likelihood	***Insignificant 1***	***Minor 2***	***Moderate 3***	***Major 4***	***Catastrophic 5***
A (almost certain)	High	High	Extreme	Extreme	Extreme
B (likely)	Medium	High	High	Extreme	Extreme
C (possible)	Low	Medium	High	Extreme	Extreme
D (unlikely)	Low	Low	Medium	High	Extreme
E (rare)	Low	Low	Medium	High	High

Ecological Risk Assessments

The initial application of risk assessment was in non-environmental fields (e.g., finance and insurance); the first environmental risk assessments were spurred by the Piper Alpha oil platform disaster of 1988, as well as the increasing attention to the threat of toxic chemicals on human health (Committee on the Institutional Means for Assessment of Risks to Public Health 1983). Ecological risk assessment began, informally, with the US *National Environmental Policy Act* (NEPA), which required the preparation of an environmental impact statement for any action that significantly effects the environment. Many of the early ecological risk assessment methods were based on those from chemical risk assessments, a model established by the US National Research Council in 1983 (Hayes 1997).

In 1990, the US Environmental Protection Agency (EPA) Science Advisory Board released *Reducing Risk: Setting Priorities and Strategies for Environmental Protection*, which recommended that the US EPA should consider the reduction of ecological risk as important as the reduction of human risk (Byrd and Cothern 2000). As a result, the US EPA used components from the chemical risk assessment framework to develop their *Framework for Ecological Assessment* (1992), which focused on ecosystem effects. While a first step in providing a risk assessment method for aquatic biosecurity, this framework is insufficient to serve as a standard framework to assess the risk from the wide range of environmental hazards. For example, the considerations for an aquatic biosecurity risk assessment will be different from many standard ecological risk assessments, e.g., highway construction through wetlands or nonpoint source air pollution (Hayes 1997).

In 1995, the Council of Standards Australia and Council of Standards New Zealand developed the first edition of the risk management standard AS/NZS 4360, which has been subsequently used by the Australian government (e.g., Commonwealth of Australia 1996) and forms the basis for the first international risk management standard, AS/NZS ISO 31000:2009 (Standards Australia 2009). AS/NZS ISO 31000:2009 are generic guidelines intended for adaptation based on the relevant objectives and projects across disciplines. In 2007 (updated in 2009), the Australian Department of Agriculture, Forestry and Fisheries (DAFF) released their Import Risk Analysis Handbook. The purpose of this handbook is to prevent or control the intentional import and subsequent establishment or spread of pests and diseases that could cause significant harm to people, animals, plants and other components of the environment (DAFF 2009), given the relevant obligations under the WTO SPS Agreement and risk assessment standards under the IPPC and OIE.

Aquatic biosecurity risk assessment frameworks

The differences between aquatic biosecurity and ecological risk assessments result from several characteristics of biological hazards: (1) biological stressors reproduce and multiply, which can lead to a time lag between when a species is introduced and when it imposes the full impact; (2) biological stressors disperse in a variety of ways that are more difficult to predict than chemical dispersal; (3) it's difficult to predict biological interactions with biotic and abiotic ecosystem parts;

and (4) biological stressors have the potential to evolve and adapt (Suter 1993, Simberloff and Alexander 1994, Stohlgren and Schnase 2006).

Although most biosecurity risk assessments contain the same general components (likelihood and consequence), there is no standardized framework: each assessment requires decisions regarding the drivers, focus, likelihood and consequence considerations, and type of analysis (Campbell 2009).

- Drivers. The assessment can be quarantine-driven or impact-driven. A quarantine-driven assessment focuses on the likelihood component (assumes impacts are significant) and uses likelihood as the indicator of risk. Impact-driven assessments use a combination of likelihood and consequence to determine the highest risks (Campbell 2009).

- Focus. The focus of the assessment can be at the species, vector or pathway level. Species-focused risk assessments may be applied to intentional or unintentional introductions or translocations to help identify high risk ANS (Azmi 2010, Hewitt et al. 2010). Vector-focused risk assessments identify, within a vector (e.g., vessels or aquaculture gear), which activities or objects pose a risk. Pathway-focused risk assessments compare relative risk between vectors or "nodes" (e.g., ports, harbours or in-water cleaning stations) (Campbell 2009, Azmi 2010).

- Likelihood. To estimate the likelihood component, the assessment can choose from several methods, such as environmental matching or species-specific assessments. Environmental matching methods compare environmental conditions (e.g. temperature, salinity) between the donor and recipient ports or locations, under the assumption that high similarity will indicate a greater chance of successful organism establishment or spread (Hayes 2003). However, which environmental characteristics make the best predictors of these events is relatively unknown (Mack et al. 2000) and remains elusive despite significant effort to determine global characteristics of invasive species (Enserink 1999, Williamson 1999, Kolar and Lodge 2000). The species-specific approach selects a species or suite of species to assess via comparisons of the species' life history and physiological traits to the environmental conditions in the recipient port or location (e.g., Kolar and Lodge 2002, Clarke et al. 2004, Gollasch 2006, Bomford et al. 2010). The environmental matching approach requires less data but can have a less conservative outcome than a species-approach (resulting in a finding of artificially low risk) because apparent environmental differences may be less than the data suggest or may not actually present a barrier to a species' successful introduction, establishment or spread (Hayes 2003, UNEP/MAP-RAC/SPA 2008). For example, the water hyacinth, *Eichhornia crassipes*, was introduced to ornamental ponds in Florida, but has spread beyond what its native range (Amazon basin) would suggest, throughout much of the southeast US and as far north as the top of California's central valley (Mack 1996). Species-based assessments require greater amounts of data (e.g. species distributions, reproductive characteristics, physiological constraints and environmental preferences) and often have more conservative outcomes than an environmental-matching approach (resulting in a finding of artificially high risk) (UNEP/MAP-RAC/SPA 2008).

- <u>Consequence</u>. Of the two risk components (likelihood and consequence), more effort has been focused on the likelihood component, with consequence assessment receiving relatively little attention (Parker et al. 1999, Hayes et al. 2004). This is partially due to the scarcity of ANS impact data, and, when available, its existence in a form inaccessible or not easily digested by resource managers (Byers et al. 2002). Within consequence assessment, there are several major considerations: the choice of values to assess; how to measure the impact for each value and categorize the consequence to these values; the choice of assessment methodology; the influence of risk perception; the use of various information types; the management of uncertainty (ambiguity, knowledge gaps, systematic and random measurement error, and variability); and the use of precaution to address this uncertainty (Campbell 2008, 2009).

- <u>Type</u>. The assessment can be quantitative, qualitative or semi-quantitative (Hayes 1997).

    - Quantitative assessments place numerical probabilities or descriptors on the elements of the risk assessment. Subsequently, they have been viewed as more objective and accurate, with less potential for misinterpretation (Fiorino 1989). However, while these beliefs are potentially valid with sufficient information, they require large amounts of data, financial and other resources (Campbell 2009). While occasionally used in aquatic biosecurity (e.g., Stone et al. 1997, Hayes and Hewitt 2000), there is generally not enough information or resources to complete a quantitative analyses for ANS (Ricciardi 2003). While possible to complete a quantitative analysis with insufficient information, the results may not justify the effort (Morgan and Henrion 1990).

    - Qualitative assessments use categorical descriptors such as "low", "medium" and "high" to determine comparative levels of risk. They are relatively inexpensive, quick, simple, feasible (when little data is available), and more easily interpreted by those without risk assessment experience (Byrd and Cothern 2000). However, they are sometimes criticized for containing greater uncertainty due to the influence of subjective judgment and perception, and leading to difficulty in making management decisions, especially when resource trade-offs are necessary (Hayes 1997, Cox 2008).

- Semi-quantitative assessments combine qualitative and quantitative data to create categorical descriptors of likelihood (with associated probabilities expressed as a percentage) and consequence (with qualitative or quantitative descriptions of each level) to determine risk (Campbell and Gallagher 2007). Semi-quantitative assessments often use quantitative data, but represent the data and outcomes in a qualitative manner. Qualitative data can also be added to the assessment and would include situations where data has been captured and combined with stakeholder and expert perceptions and empirical data (Hewitt et al. 2010).

Uncertainty and false certainty

Uncertainty constitutes an inherent component of risk given the unknown characteristics of a threat and the associated predictive efforts of assessing the risk of that threat (Morgan and Henrion 1990, Yates and Stone 1992). Uncertainty is a concept with as many definitions as disciplines in which it occurs. Both uncertainty and the related fields of risk assessment lack a shared definition of uncertainty and related terminology (Walker et al. 2003). However, when describing the typology of uncertainty, a common delineation occurs between epistemic and ontological varieties (Walker et al. 2003, Cooney and Lang 2007). Epistemic uncertainty stems from a lack of knowledge and can be ameliorated via additional research or similar efforts (Walker et al. 2003). Ontological uncertainty stems from the inability to fully describe a variable and complex environment and cannot be eliminated (Walker et al. 2003).

In an impact assessment context, these two types lead to several specific sources of uncertainty, including: (1) knowledge gaps; (2) systematic and random measurement error (e.g., flawed measurements and uncertain or inappropriate models); (3) indeterminacy (unavoidable, stochastic behaviour between hazard and impact); (4) variability (the variety of impacts that the same hazard can have in different locations of time and space); and (5) ambiguity (e.g., different interpretations of the same data set) (Byrd and Cothern 2000, Klinke and Renn 2002).

Using the Walker et al. (2003) definition of uncertainty, “any deviation from the unachievable ideal of completely deterministic knowledge of the relevant system”, another source of uncertainty is the ‘false uncertainty’. False certainty stems from interpretation error where the assumption of an outcome is overstated or overly simplified, such as the interpretation of a statistically non-significant result as inferring “no impact”, despite low power (i.e., a low probability of detecting an effect, given there is one). In an impact assessment context, this false certainty can obscure ANS effects on a native species, community or ecosystem due to insufficient sample or effect size or inappropriate experimental design, leading to Type II errors.

Uncertainty and risk management

Both reducible and irreducible forms of uncertainty have implications for risk management (Smithson 2008). Epistemic uncertainty, which can often be identified and addressed through additional research or including alternative forms of knowledge, undoubtedly presents a challenge to risk management (Walker et al. 2003). In an ANS risk assessment context, this includes understanding what forms of uncertainty exist and to what extent, finding ways to reduce this uncertainty that are acceptable to expert and stakeholder groups as well as international trade bodies such as the WTO, and finally, implementing these methods using the (often limited) resources available to aquatic biosecurity agencies. However, in situations in which uncertainty cannot be reduced, for reasons of an ontological or practical nature (such as time or other resource constraints), risk management is faced with a more difficult task. That is, making decisions despite extensive knowledge gaps, in which subjective expert and stakeholder judgment play a key role and controversy or disagreement are common (a state described by Kasperson as 'deep uncertainty'; Kasperson 2008).

Both expert judgment and precaution have been proposed as methods to mitigate uncertainty, with challenges associated with each (Stirling and Gee 2002, Teck et al. 2010). Expert judgment is used in many environmental contexts with general success in minimizing the effects of uncertainty in reaching a decision (e.g., Meyer and Booker 1990, Campbell and Gallagher 2007, Therriault and Herborg 2008, Donlan et al. 2010, Teck et al. 2010). However, it is susceptible to subjective factors that influence the cognitive decision-making process (i.e., heuristics and biases) (Smithson 2008). Precaution is a tool used in environmental management, positing that the presence of uncertainty shall not prevent measures to prevent or minimize significant harm (Peel 2005). The Convention on Biological Diversity (CBD) not only directs Members to use precaution, but also to "prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species" (CBD Secretariat 1992). This suggests that uncertainty within ANS risk assessment should be managed in a precautionary manner, i.e., assume "guilty until proven innocent" and implement measures to mitigate the threat (Campbell et al. 2009). However, the application of precaution is fraught with criticisms and viewed as "unscientific" by many individuals and regulating bodies, including the WTO (Tucker and Treweek 2005, Peterson 2006). In addition, due to the context-dependent application of precaution, there is no common prescription for how and when precaution should be applied (Peel 2005). Understanding when the use of expert judgment and the application of precaution are appropriate, and how they should be incorporated into the risk assessment process, presents significant challenges (Peel 2005).

Challenges for aquatic biosecurity risk assessment

In response to the general recognition of gaps, redundancies and inconsistencies in terminology, content, and process within measures on the prevention, early detection, eradication and control of invasive species, the CBD Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA) reviewed relevant national, regional and international measures (Subsidiary Body on Scientific Technical and Technological Advice 2001). The review found gaps in regulatory frameworks for animals that are not pests of plants (or, are pests but of marine plants), as well as for the ballast water and hull fouling vectors (Subsidiary Body on Scientific Technical and Technological Advice 2005). The report also found varying levels of understanding, interpretations and application of precaution in the resolution of uncertainty (Shine 2006). This review of the regulatory framework suggests the presence of gaps and/or inconsistencies in aquatic biosecurity measures aimed at the ballast water and hull fouling vectors, and specifically, aquatic biosecurity risk assessment frameworks, at the national, regional and international level.

Challenges for aquatic biosecurity consequence (impact) assessment

While a relatively large number of articles have been published on the impacts of ANS, literature providing an impact assessment framework applicable in a marine biosecurity risk assessment context is lacking (Parker et al. 1999). That is, frameworks that can provide broad descriptions of varying impacts for a broad suite of species, account for uncertainty, and are easily integrated and prioritized to inform policy development or management action, are scarce (Parker et al. 1999). While several articles provide a framework to categorize a subset of potential impacts, they are often limited taxonomically, spatially or by impact type. For example, Ruiz et al (1999) completed an impact assessment that, while comprehensive in its treatment of nine types of environmental impacts (competition, habitat change, food-prey, predation, herbivory, hybridization, parasitism, toxicity and bioturbation) by 196 ANS in Chesapeake Bay, did not include other core values.

Aims, hypotheses and thesis structure

In the face of a growing ANS threat, scientists and managers look to risk management strategies to minimize or eliminate this threat (Hewitt and Hayes 2002). However, these are limited by the presence of significant gaps and inconsistencies in existing national, regional and international aquatic biosecurity risk assessments, as well as uncertainty within the knowledge and process used in the assessment (Dahlstrom et al. 2011). Policy and regulatory bodies require assessors to make decisions of consequence in the presence of significant uncertainty and scarce resources (resources necessary to ameliorate that uncertainty through additional research) (Hattis and Anderson 1999). As such, this thesis intends to identify these broad gaps and inconsistencies, then focus on the identification of uncertainty, how uncertainty affects expert judgment, methods to reduce this uncertainty (when possible) and (when not possible) outline measures for estimation of ANS consequence that are transparent and allow for varying degrees precaution. Understanding the implications of uncertainty, as well as effective measures to mitigate these implications, will lead to more successful risk-based management of ANS. This chapter reviews the increasing threat of ANS impacts juxtaposed against the uncertainty surrounding risk-based decision making. These considerations provide the foundation for the aims and structure of the following chapters.

To this end, in Chapter 2 (publication provided in Appendix A) I use thematic analysis to compare the various components of biosecurity risk assessment frameworks for 14 national, regional and international biosecurity instruments, including the risk definition, risk assessment principles, terminology, information type, likelihood and consequence considerations, core values and subcomponents, and mention of precaution and uncertainty. Based on the weaknesses discussed above, I expected the review to find variety in the content and sufficiency of both the descriptions and prescriptive directions for many of the framework components. The outcomes of this review are used to identify similarities, differences, and deficiencies in the frameworks and from these provide recommendations to improve the content and process of aquatic biosecurity risk assessment.

Despite significant sources of uncertainty in ANS management, such as knowledge gaps, confusing terminology and spatial and temporal variability, the review of biosecurity risk assessment frameworks found limited mention of how to address and mitigate this uncertainty. Where the frameworks do address uncertainty, they offer the use of expert judgment (often via the Delphic process) and precaution as potential solutions, but with limited guidance as to their implementation. As such, in Chapter 3 I survey ANS science and management experts to identify where and in what forms uncertainty exists and how it can be best addressed in a consequence assessment context. I also hold a 'mock' consequence assessment exercise using a subset of ANS to both determine the effects of uncertainty on consequence estimates, and test the functionality of a Delphic process in aiding such decision making under uncertainty. Experts were challenged to identify significant sources of uncertainty, particularly due to knowledge gaps, with moderate endorsement of precaution and limited endorsement of alternative information sources (such as using observational information) to make an assessment. I also anticipated that the Delphic process would decrease uncertainty. Based on the outcomes of the survey and consequence assessment, I present methods to facilitate the completion of consequence (and hence, risk) assessments in the presence of reducible and irreducible uncertainty in a manner that is supported by the experts providing the input, and thus in accordance with the science-based mandates of the WTO.

Chapter 4 explores the effect of uncertainty on the cognitive decision making process in a biosecurity risk assessment context via the relationship between uncertainty and consequence estimates observed in Chapter 3. This chapter also includes investigations into the effect of several other heuristics and biases affecting the perception of a species consequence on core values. Given that both normative decision theories and the use of precaution posit that decision making under uncertainty should maximize expected value via erring on the side of caution to avoid making Type II errors, it was anticipated that assessors would assign greater consequence when faced with irreducible uncertainty. However, descriptive models repeatedly find that uncertainty leads to the use of cognitive heuristics and biases. As such, I anticipated that several biases would differentially affect the estimates for several species, such as those from a genus well-known for severe impacts (e.g., *Caulerpa*) or those having economic impacts (e.g., *Bonamia ostreae*) as opposed to environmental impacts. Understanding the cognitive processes in, and influences on, expert decision making process will provide for a more transparent risk assessment process and facilitate the development of measures to mitigate or account for these influences (where appropriate).

In addition to the traditional forms of uncertainty addressed in Chapters 3 and 4, the false certainty arising from low-powered statistical analyses that find an insignificant statistical effect may have severe implications for biosecurity risk assessment and management. Chapter 5 uses algal and crustacean ANS impact studies to determine the prevalence of low power in risk and impact-based research. Given the traditional use of a null hypothesis that assumes no difference between treatments (central to significance testing methods), combined with the low acceptable rate of Type I errors (“false alarms”), I anticipated finding low power in many of the studies. Due to the implications of this potential outcome (i.e., high rates of Type II errors or “missing” an impact), I provide alternative impact assessment methods that incorporate the pre-determined acceptable level of risk and associated costs of each error type. These methods help align the respective needs and outcomes of biosecurity research and management, as well as improving the communication between the two sectors.

Chapter 6 synthesizes the outcomes of the previous chapters and provides implications for management efforts. It underscores the importance of a framework that can provide direction despite uncertainty and also incorporate policy mandates (such as acceptable level or risk) due to recent economic and political factors contributing to an increase in potential transfer of ANS. Present and growing global factors such as military expansion, local energy-related development, the global financial crisis, and regional trade agreements all lead to increased connectivity and hence increased risk of species introductions. The continued increase in the magnitude of globalization underscores the importance of developing a comprehensive risk assessment framework that can operate despite uncertainty. These factors highlight the importance of not only integrating scientific data when completing risk assessment, but also 'non-scientific' (e.g., economic and political) considerations into decisions of risk. I conclude with a model that provides transparent guidance for assessing consequence given both available and scarce information.

CHAPTER 2. A REVIEW OF INTERNATIONAL, REGIONAL AND NATIONAL BIOSECURITY RISK ASSESSMENT FRAMEWORKS

Published and provided in Appendix A:

Dahlstrom, A., C. L. Hewitt, and M. L. Campbell. 2011. A review of international, regional and national biosecurity risk assessment frameworks. Marine Policy 35:208-217.

Dahlstrom, A (60%), Hewitt, CL (20%), Campbell, ML (20%)

Introduction

Globally over 1781 nonindigenous species are known from marine or brackish waters (Kolar and Lodge 2000, Hewitt and Campbell 2008), and over 179 in the freshwater systems of Europe and the Laurentian Great Lakes alone (Elvira 2001). To mitigate potential impacts caused by these species (see Chapter 1), biosecurity efforts often focus on prevention. Risk assessment is a proven pre-border management tool that facilitates efficient and effective ANS management by allowing managers to determine the relative significance of various species and vectors, establish an Acceptable Levels of Risk (ALOR), and Protection (ALOP), and prioritize use of resources accordingly.

While biosecurity risk assessment is recognized as a worthwhile exercise that can enable efficient prioritization of resources to high-risk hazards, it is still a young and rapidly-evolving field, with few risk assessment frameworks specific to aquatic biosecurity. Of the two components (determining likelihood and consequence; Chapter 1), most effort has been focused on determining likelihood; consequence assessment for aquatic biosecurity risk assessments has received relatively scant attention (Parker et al. 1999). Different consequence assessment models have included core value categories in different combinations and have used different definitions, dependent upon the context. Some models are limited to environmental consequences such as biodiversity (Orr et al. 1993), others economic (Pimentel et al. 2000), and others a subset of the five core values (Fofonoff et al. 2003, Campbell 2005, 2008).

Many national and regional biosecurity policies that may affect trade (e.g., the development or review of import health standards, surveillance programmes, and incursion responses) are underpinned by risk assessment, as required by the WTO Sanitary and Phytosanitary (SPS) Agreement. Article 5 of the SPS Agreement on SPS measures[4] specifies obligations WTO members have with respect to SPS-mandated risk assessments, including a science-based methodology and minimizing negative impacts to trade (WTO 1995, Campbell 2001). The WTO SPS Agreement identifies several standard-setting bodies that have established risk assessment frameworks, on which WTO members should base their own risk assessments, including the International Plant Protection Convention (IPPC), the FAO/WHO Codex Alimentarius Commission (focused on food safety), and the World Organization for Animal Health (OIE). In addition to the WTO/IPPC/OIE, other entities with risk assessment-based biosecurity policies include the Convention on Biological

[4]An SPS measure is any measure designed to protect human or animal life from risks arising from additives, contaminants, toxins or disease-causing organisms in their food; to protect human life from plant- or animal-carried diseases; to protect animal or plant life from pests, diseases, or disease-causing organisms; or to prevent or limit other damage to a country from the entry, establishment or spread of pests (WTO 1998).

Diversity, International Maritime Organization, the Antarctic Treaty System, the North America Free Trade Agreement, the Asia-Pacific Economic Cooperation, the South Pacific Regional Environment Programme, the Barcelona Convention, countries such as Australia, New Zealand and the United States.

Review of these biosecurity risk assessment frameworks identified significant differences in their content and process requirements (Dahlstrom et al. 2011; Appendix A), which is due in part to the foundation of their framework: many were developed based on risk assessment models in other fields (e.g., toxicology and economics). Given the increasing rates of ANS transfer and the reliance on risk assessment to manage and regulate ANS, understanding and reconciling these differences is imperative (Gollasch 2006). The insight generated by this review will help facilitate harmonization of the frameworks and increase the effectiveness of domestic, regional and international risk assessment activities, minimizing the risk of species introduction, the impacts to trade and the resources necessary for the assessment, as well as improving future biosecurity policy efforts.

To this end, this chapter undertakes a review that examines the risk assessment frameworks adopted by a suite of international, regional and national agreements pertaining to biosecurity. The objectives of this review are to: (1) determine the number and scope of aquatic biosecurity risk assessment frameworks within the instruments[5]; and (2) determine and compare the associated principles (e.g., use of precaution and uncertainty, the components of each risk assessment and the values used in each framework's consequence assessment) for the national, regional, and international risk assessment frameworks. It is anticipated that that the outcomes will contribute to the development of risk assessment methodology by describing the current status of biosecurity risk assessments via the objectives above, identifying several challenges and weaknesses in the current aquatic biosecurity frameworks, and making recommendations for improvements to aquatic biosecurity risk assessment and policy.

Methods

This review focused on risk assessment frameworks (and specifically, the consequence assessment component) for a suite of national, regional, and international instruments and their relevant policies. Initially, this review focused on risk assessment frameworks for unintentional introductions of ANS via biofouling. However, due to a dearth of these types of risk assessment frameworks, the

[5] For this review, an 'instrument' consists of agreements, conventions or policies that describe the process and/or content of a risk assessment for the relative organization.

approach broadened to a review of all biosecurity risk frameworks, taking a more narrow focus (e.g. marine biosecurity) where appropriate. For each instrument reviewed, the framework most relevant to unintentional introductions of ANS is presented.

Original risk assessment sources are used for each source (e.g. published risk assessment manuals, risk analysis handbooks, or international standards). Where published articles or other review documents existed and assisted with the review, they were also included.

The instruments were selected on the basis of relevance at the national, regional, or international stage. Seven international instruments were reviewed: the WTO; its three standard-setting bodies (International Plant Protection Committee (IPPC), World Organization for Animal Health (OIE), and the Food and Agricultural Organization (FAO)); the Convention on Biological Diversity (CBD); the International Maritime Organization (IMO); and the Antarctic Treaty System (ATS). Four regional instruments were reviewed: the North America Free Trade Agreement (NAFTA); the Asia-Pacific Economic Cooperation (APEC); the South Pacific Regional Environment Programme (SPREP); and the Barcelona Convention (BC). National instruments focused on Australia, New Zealand and the United States (US). These instruments are discussed briefly below.

- The WTO is an international trade organization with numerous agreements that aim to facilitate and promote free trade (e.g., the SPS Agreement) and must be ratified by current members and any new member states. It was established in 1995 to supersede the General Agreement on Tariffs and Trade (GATT), which until then had been the sole trade agreement since 1947. The risk assessment framework used in this analysis is that established in SPS Agreement Article 5, which was established with the WTO in 1995 to ensure measures for the protection of human or animal life do not constitute arbitrary or unjustifiable barriers to trade (WTO 1995).

- The International Plant Protection Convention (IPPC) is an international convention under the United Nations Food and Agricultural Organization (FAO) that aims to prevent the spread and introduction of pests of plants and plant products. The IPPC was established in 1952 and revised in 1997 to reflect the principles of the WTO, of which it is a designated standard-setting body. The risk assessment framework used in this analysis is that established in the International Standards for Phytosanitary Measures (ISPMs) 2, 5 and 11 (IPPC Secretariat 2002).

- The World Organization for Animal Health (OIE) is an international organization formed in 1924 to promote world animal health via a focus on diseases and pathogens, and is designated as an international standard setting body by the WTO. The risk assessment framework used in this analysis is that established in the *OIE Aquatic Animal Health Code* (OIE 2009a).

- The Food and Agriculture Organization (FAO) is a United Nations organization that was created in 1945 to reduce world hunger and is designated as an international standard setting body by the WTO. The FAO provides policies and information for all nations to use. The risk assessment framework used in this analysis is that established in *Understanding and applying risk analysis in aquaculture: a manual for decision-makers* (Arthur et al. 2009).

- The Convention on Biological Diversity (CBD) is a United Nations Environmental Programme convention that entered into force in 1993 to conserve biological diversity, sustainable use of its components, and the fair and equitable sharing of the benefits from use of genetic resources. The risk assessment framework used in the analysis is that described in decisions by Conferences of the Parties to the CBD (CBD Secretariat 2000, 2002, 2004, 2008).

- The International Maritime Organization (IMO) is a United Nations agency that was established in 1948 to promote safe, secure and efficient shipping on clean oceans (IMO 2010). The risk assessment framework used in the analysis is that described in the *International Convention for the Control and Management of Ships' Ballast Water and Sediments*, which was adopted in 2004 but has not yet entered into force (IMO 2010).

- The Antarctic Treaty System (ATS) is a system of international policy designed to maintain Antarctica as a non-military scientific preserve. The ATS consists of the original Antarctic Treaty (established in 1961) and many additional agreements and conventions, including the protocol used in the analysis (the *Protocol on Environmental Protection to the Antarctic Treaty* (Madrid Protocol)) (British Antarctic Survey 2007). This protocol entered into force in 1998 and includes direction on non-native species and environmental impact assessments.

- The North America Free Trade Agreement (NAFTA) is a trade agreement established in 1994 that created a free trade agreement between Canada, the United States and Mexico. The Commission for Environmental Cooperation (CEC) was established to promote environmental protection while increasing trade and economic links between the three countries. The risk assessment framework used in the analysis is that described in the CEC's *Risk Assessment Guidelines for Aquatic Alien Invasive Species* (Mendoza et al. 2009).

- The Asia-Pacific Economic Cooperation (APEC) is a regional agreement established in 1989 to enhance economic growth and prosperity in this region. Within APEC, the Marine Resource Conservation Working Group (MRCWG) was established in 1990 to promote sustainable and efficient use of the marine and costal environment while supporting free trade and investment. The risk assessment framework used in the analysis is that described in the MRCWG's *Regional Risk Management Framework for APEC Economies for Use in the Control and Prevention of Introduced Marine Pests* (Asia-Pacific Economic Cooperation 2005).

- The South Pacific Regional Environment Programme (SPREP) is a regional agreement established in 1982 to protect and improve the environment in the Pacific Islands region. The risk assessment framework used in the analysis is that described in the *Regional Strategy on Shipping-Related Introduced Marine Pests in the Pacific Islands Region* (SPREP 2006).

- The original version of the *Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean* (Barcelona Convention, BC) was developed in 1976 to assess and control pollution and improve the environment of the Mediterranean Sea. This was amended in 1995 and now includes 22 Mediterranean countries as Members and several binding protocols, specifically the *Specially Protected Areas and Biological Diversity Protocol* (SPA/BD Protocol). The risk assessment framework used in the analysis is that developed in the context of the SPA/BD Protocol and Barcelona Convention, the *Guide for Risk Analysis assessing the Impacts of the Introduction of Non-indigenous Species* (UNEP/MAP-RAC/SPA 2008).

- Australia, New Zealand, and the United States are nations that have a risk assessment framework that will be included in the analysis. The risk assessment framework used in the analysis for Australia is the Australian Department of Agriculture, Fisheries and Forestry's *Import Risk Analysis (IRA) Handbook*[6] (DAFF 2009). The risk assessment framework used in the analysis for New Zealand is the Ministry of Agriculture and Fisheries Biosecurity New

[6] Each of these countries has adopted ballast water management (BWM) policies that include risk assessment as a component, more notably the Australian Quarantine and Inspection Service Ballast Water Decision Support System. However, these individual BWM policies were not included in the review as it was considered more appropriate to choose the national instruments that focus on policies and procedures of the risk assessment process that can be adapted to a specific species suite, vector or pathway (e.g., biofouling) and give attention to consequence assessment (BWM risk assessments typically give more weight to the likelihood assessment).

Zealand *Risk Framework* (Biosecurity New Zealand 2006). The risk assessment framework used in the analysis for the United States is US Aquatic Nuisance Species Task Force *Risk Framework* (Risk Assessment and Management Committee 1996).

These instruments include non-binding agreements, conventions and national policies. The conventions were developed by international bodies to formally adopt specific principles or objectives. A member of the international body must ratify (become a signatory to) the convention if it is to take force and become binding for that member state. For example, the United States is a Member of the United Nations, but has not ratified the CBD so is not a Party to this convention or bound by its principles.

Preliminary Analyses

To ensure consistency, each instrument review gathered the same set of information: instrument name, description, purpose/aim, background, standard setting body and relevant policy, risk definition, risk assessment framework (general or aquatic biosecurity), information requirements for risk assessment, the use of precaution and uncertainty within the risk assessment, endpoint likelihood assessment, and consequence assessment (for general or aquatic biosecurity risk assessment) (Table 2.1).

Table 2.1. Description of components included in the preliminary analysis.

Components of the preliminary analysis	Description
Instrument name and Description	Provides the name and type of the instrument (e.g. "regional trade organization" or "international environmental organization")
Purpose/aim	Provides the purpose or aim of the instrument and, where appropriate, the purpose/aim of any significant bodies or policies directly responsible for risk management (or, where available, those responsible for ANS risk management)
Background	Provides the force (i.e., nonbinding or binding) and principles of the instrument
Risk Definition	Define the method of risk analysis and/or assessment (differences exist between these terms, as discussed in Chapter 1)
Risk Assessment Framework	Provides a review of risk assessment frameworks, most relevant to unintentional introductions of ANS, that includes terminology used and the type of risk assessment. Where no aquatic biosecurity risk frameworks existed, a general biosecurity risk framework was reviewed.
Information requirements for risk assessment	Identifies the forms and sources of information and data allowable and/or recommended for use in the risk assessment

Table 2.1 cont.

Components of the preliminary analysis	Description
Precaution within Risk Assessment	Reviews any mention or discussion of precaution, the Precautionary Principle, or the Precautionary Approach by the instrument, or where present, in the risk assessment framework itself
Uncertainty within Risk Assessment	Reviews any mention or discussion of uncertainty by the instrument, or where present, in the risk assessment framework itself
Endpoint Likelihood Assessment	Provides additional detail on the methods provided in the risk assessment for identifying the probability of the risk
Consequence Assessment	Provides additional detail on the methods provided in the risk assessment for identifying the effects of the risk (including the core values identified). Occasionally the consequence component is embedded in the "risk assessment framework" section, in which case the reader is referred there. Where additional, stand-alone detail exists, however, it is included in this section. Several of the consequence subcomponents were terrestrial focused, which were adapted for an aquatic context where appropriate (e.g., impacts to agriculture or crops in the IPPC assessment were categorized as impacts to aquaculture and fisheries)

Thematic analysis

To analyse the preliminary data, content analysis was used to identify recurring themes that became categorical factors. These themes were used to determine basic (in)consistencies between each instrument (Patton 2002). After the content analysis, it was apparent that the risk assessments either agreed or differed according to several factors: force of the instrument, principles of the risk assessment, terminology for the species under assessment, type of risk assessment, information type, the use of precaution, factors in determining likelihood, and considerations determining consequence and core values included in the consequence assessment (Table 2.2).

Table 2.2. Description of components included in the thematic analysis.

Components of the thematic analysis	Description
Force of the instrument	Force of the instrument was binary; the instrument's measures and risk assessment were "binding" or "non-binding".
Principles of the risk assessment	Principles of the measures and/or risk assessments for each framework included a subset of the following: transparent, national sovereignty, science-based risk assessment, international cooperation, trade facilitation, document uncertainty, and being WTO-compliant.
Terminology for the species under assessment	Species terminology was compared via content analysis, i.e., the description of the species posing the threat was coded as one of two possible categories: general (e.g. pathogen or pest) or specific to invasion biology (e.g., nonindigenous or invasive).
Type of risk assessment	Type of risk assessment was qualitative, quantitative, semi-quantitative or a combination.
Information type	Information type included a subset of the following: scientific literature ("science-based" risk assessments included here), expert opinion, stakeholder (including industry) opinion, anecdotal reports, experience-based input, and other (which is not a bin for specific information types, but reflects the non-specific mention of "other" or "variety of" sources by the risk assessment).
Use of precaution	The mention of precaution was classified as yes (specific mention of precaution), not explicitly (no specific mention of precaution, but may be relevant via provisional measures, etc) and no (no mention of precaution or related measures).
Considerations determining consequence	Considerations for determining consequence are listed comprehensively, with overlap noted.
Core values	Core values included a subset of the following: biological, ecological, economic, environmental, socio-economic, economic, social, cultural, socio-political, and human health. Biological, ecological, and environmental were combined under the heading "environmental". Socio-economic was scored as both "social" and "economic," and socio-political was scored as "social".
Consequence direction and subcomponents	The direction of consequences was also noted, as either negative (N), both negative and positive (Either) or did not specify (DNS). Finally, a comprehensive list of consequence subcomponents was compiled.

Results

Of the 14 instruments reviewed, 6 had risk assessments specific to aquatic biosecurity: the OIE, IMO, FAO, NAFTA, BC and US. Although SPREP and APEC did not have official risk assessment frameworks, they did provide considerations for an aquatic biosecurity risk assessment framework. A summary of the following characteristics are in Table 2.3: force of the instrument; principles of the risk assessment; use of precaution; type of risk assessment; and information type. Each of the instruments contained multiple principles of the risk assessment, information types, and core values; as all principles/types/values were recorded, the total for these areas will not sum to fourteen. These characteristics are displayed by instrument in Table 2.4.

Table 2.3. Summary of results from: national, regional and international risk assessment framework analysis.

Factor		Total(%)
Force of instrument	Binding	11(79%)
Risk assessment principles	Transparency	6(43%)
	National sovereignty*	6(50%)
	Science-based	11(79%)
	International cooperation	9(64%)
	Facilitate trade	7(50%)
	Document uncertainty	8(57%)
	WTO-compliant**	5(39%)
Endorses the use of precaution	Yes	8(57%)
	Potentially, through provisional agreements	3(21%)
	No	3(21%)
Risk assessment type	Qualitative only	0(0%)
	Quantitative only	0(0%)
	Qualitative and quantitative	4(29%)
	Qualitative, semi-quantitative, and quantitative	5(38%)
	Did not specify	5(38%)
Information type	Scientific literature	10(71%)
	Anecdotal	3(21%)
	Experience-based	3(21%)
	Expert judgment	3(21%)
	Stakeholder	2(14%)
	Other	2(14%)
Core values	Environmental	13(93%)
	Economic	12(86%)
	Social	9(64%)
	Human health	6(43%)
	Cultural	5(36%)

Table 2.4. Summary of findings from risk assessment framework review (●=condition met; NE=not explicitly; NA=not applicable due to lack of risk assessment framework; N=negative consequences, E=Either, DNS=did not specify; G=general, I=invasion biology, A=aquatic invasion biology). AB specific indicates whether the risk assessment is specific to aquatic biosecurity. *The three national frameworks excluded to avoid redundancy. ** WTO excluded to avoid redundancy.

		International							Regional				National		
		WTO	**IPPC**	**OIE**	**FAO**	**CBD**	**IMO**	**ATS**	**NAFTA**	**APEC**	**SPREP**	**BC**	**AU**	**NZ**	**US**
	Binding	●	●	●		●	●	●	●			●	●	●	●
	Precaution				●	●	●	**NE**	**NE**	●	●	●	●	●	**NE**
PRINCIPLES	Transparency	●	●	●			●	●	●					●	
	National Sovereignty*	●	●	●			●		●			●			
	Science-based	●	●	●	●		●	●	●			●	●	●	●
	International Cooperation	●	●	●		●	●		●		●	●		●	
	Facilitate Trade	●	●	●	●		●		●	●				●	
	Document Uncertainty		●	●	●		●	●	●			●		●	●
	WTO-compliant**		●	●					●				●	●	
RA TYPE(S)	Quantitative	●	●	●	●		●		●			●		●	●
	Semi-quantitative				●				●			●		●	●
	Qualitative	●	●	●	●		●		●			●		●	●
	AB specific			●	●		●		●	**NA**	**NA**	●			●
INFORMATION TYPE	Scientific literature	●	●	●	●		●	●	●	●		●	●	●	●
	Anecdotal				●				●						●
	Experience-based				●				●					●	●
	Expert judgment			●	●							●		●	
	Stakeholder				●							●			
	Other		●	●	●										
	Consequence direction	**E**	**N**	**N**	**N**	**N**	**DNS**	**N**	**E**	**E**	**DNS**	**N**	**N**	**N**	**E**
	Terminology	**G**	**G**	**G**	**I**	**I**	**A**	**I**	**A**	**A**	**A**	**I**	**G**	**G**	**A**

Of the risk assessment principles, science-based was the most common, followed by international cooperation (Figure 2.1). Scientific literature was the most common information type allowed in the risk assessments (Figure 2.2).

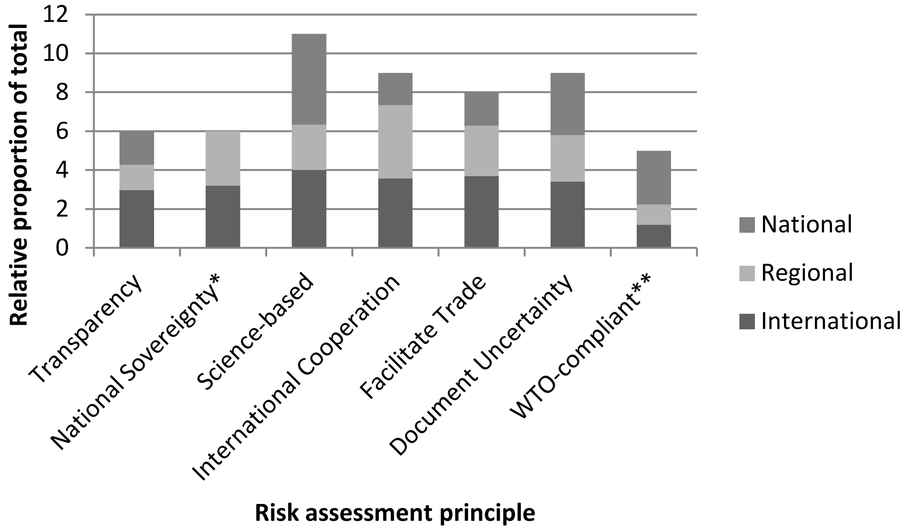

Figure 2.1. Risk assessment principles assessed from national, regional and international risk assessment framework analyses. *The three national frameworks were excluded to avoid redundancy. ** WTO was excluded to avoid redundancy.

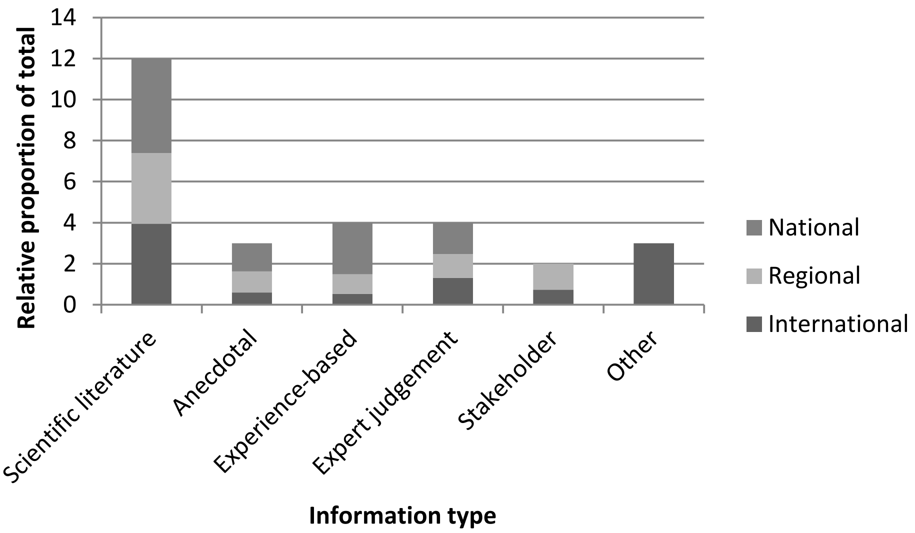

Figure 2.2. Information sources used in national, regional and international risk assessment framework analyses.

Terminology for the species under assessment

The terminology used in each assessment was classified into three categories: (1) general (e.g., pathogen or pest; IPPC, OIE, WTO, Australia and New Zealand); (2) specific to invasion biology (e.g., invasive, exotic; FAO, CBD, ATS and BC); or (3) specific to aquatic invasion biology (e.g., aquatic nonindigenous species; IMO, NAFTA, APEC, SPREP and US) (Table 2.4). Whether an instrument used (aquatic) invasion-specific terminology was substantively associated with whether the instrument had a risk assessment specific to aquatic biosecurity. None of the other frameworks with "general" terminology have a risk assessment framework specific to aquatic biosecurity, with the OIE being an exception.

Factors in determining endpoint likelihood

Although the risk assessment frameworks generally contained similar likelihood endpoints (entry, establishment, spread), one major difference was whether the biosecurity risk assessment specified the "and" condition (i.e., the assessment evaluated the cumulative likelihood at each endpoint), or used the "or" condition (the risk assessment evaluated the likelihood for one outcome, to be determined at time of assessment) (Table 2.5). Those instruments that either do not provide a method for likelihood assessment (i.e., APEC, ATS, CBD) or provide multiple options for assessing likelihood (i.e., SPREP, BC) were excluded from analysis. The IMO included evidence of prior introduction and the current distribution as specific factors for determining likelihood.

Table 2.5. Factors used to determine endpoint likelihood for national, regional and international instruments. Those instruments that either do not provide a method for likelihood assessment (i.e., APEC, ATS, CBD) or provide multiple options for assessing likelihood (i.e., SPREP, BC) were not included. Bold font differentiates those that use the "and" condition (as opposed to the "or" condition.)

Instrument	Factors in determining endpoint likelihood
IPPC	**entry, establishment and spread**
OIE	**release and exposure**
IMO	**uptake, transfer, discharge, and population establishment**
NZ	**entry, exposure and establishment**
US, NAFTA	**associated with pathway, entry, colonization and spread**
WTO	likelihood of entry, establishment or spread
FAO	impact, incursion, release, or exposure
AU	likelihood the pest or disease would enter, establish, or spread

Factors for determining consequence

While the risk assessment frameworks provided several approaches to determine likelihood, and many examples of consequence, only two frameworks provided factors to determine consequence (IMO and IPPC). The IMO recommends that for each species to consider demonstrated impacts and the strength and type(s) of ecological interactions. The IPPC recommends considering biotic factors that would affect impacts (e.g., adaptability and virulence of the pest).

Core values and subcomponents included in the consequence assessment

Of the core values included in consequence assessments, environment was the most common (mentioned in 93% of the instruments), followed by economic (86%), social (64%), human health (43%) and cultural (36%) (Figure 2.3).

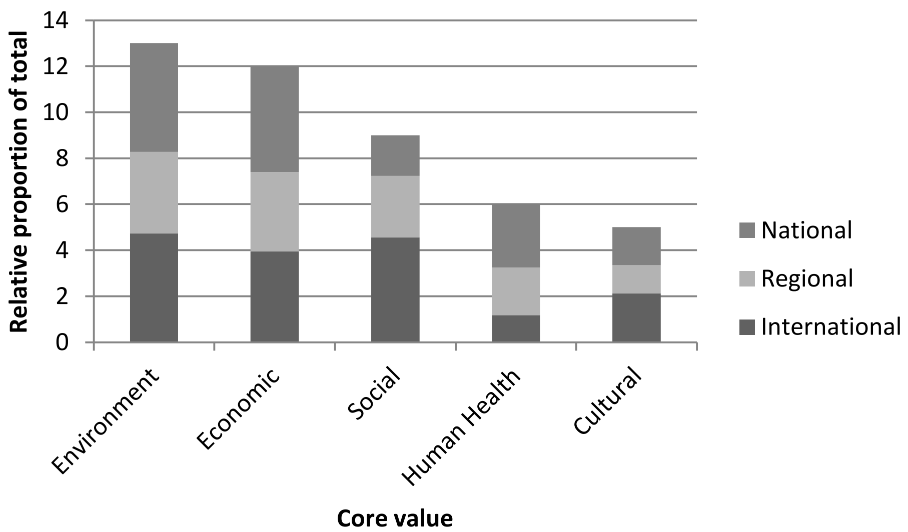

Figure 2.3. Core values that are included in each of the framework's consequence analyses.

The consequences included in the risk assessment frameworks contained a substantial amount of overlap, with each core value mentioned by at least five of the frameworks (Table 2.6). While some frameworks mentioned positive as well as negative consequences, most only listed examples of negative consequences. The APEC framework, however, provides examples of positive consequences, such as aesthetic values; new activities (e.g. fisheries and aquaculture); increased employment in marine pest management-related activities; and additional knowledge of ecosystems driven by the desire to understand the ecology of invasions.

Table 2.6. Impact categories and subcomponents included in the reviewed impact assessments. The filled circles indicate core value subcomponents; the empty circles indicate specific examples of the main subcomponent above. The filled squares indicate the instrument only indicated the core value, without mention of any subcomponents. SPREP did not list any specific subcomponents and is intentionally left blank.

	International							Regional				National		
	WTO	IPPC	OIE	FAO	CBD	IMO	ATS	NAFTA	APEC	SPREP	BC	AU	NZ	US
ENVIRONMENT		●	●	●	●		●	●	●		●	■	●	●
Pest/pathogen			●		●								●	
Pest/pathogen vector		●											●	
Biodiversity			●	●			●	●	●		●			●
Habitat loss or change				●	●			●			●		●	
Species abundance		●		●										
Decrease in keystone species		●						●					●	●
Decrease in threatened or endangered species		●					●	●			●		●	●
Toxicity													●	
Ecological interactions		●	●										●	●
Predation					○								○	
Herbivory					○									
Competition					○								○	
Hybridization					○			○					○	
Ecosystem processes		●						●					●	
Nutrient regime changes		○			○									
Hydrological cycle changes		○			○		○						○	
Food web changes				○	○									
Physical disturbance					●								●	
Effects of control measures		●						●						●
Effects of climate change on invasive species and/or impacts					●									

Table 2.6 cont.

	International							Regional				National		
	WTO	**IPPC**	**OIE**	**FAO**	**CBD**	**IMO**	**ATS**	**NAFTA**	**APEC**	**SPREP**	**BC**	**AU**	**NZ**	**US**
ECONOMIC	●	●	●	●	●	●		●	●		●	●	●	●
Production	●	●	●	●	●						●	●	●	
Infrastructure/facilities/property			●			●			●		●			
Control and management costs	●	●	●		●			●	●			●	●	●
Trade losses		●	●	●				●					●	●
Adverse consumer reaction		●	●	●										●
Natural resources				●				●			●			●
Tourism		●		●					●		●		●	
Power generation					●									
Shipping (including vessels and waterways)					●				●		●			
Fisheries		●	●	●	●			●	●		●			●
Aquaculture		●	●	●				●	●		●			●
Restoration costs		●											●	
Health care costs													●	
Ecosystem services								●						
Opportunity costs									●					
SOCIAL AND CULTURAL		●	■	●	■	■	●	●	●		●		●	●
Aesthetic				●			●				●			●
Social amenity											●		●	
Political repercussions				●										●
Objects of religious importance				●							●			
Objects of cultural significance							●	●			●		●	
Indigenous community								●					●	
Learning/research opportunities							●				●			

Table 2.6 cont.

	International							Regional				National		
	WTO	**IPPC**	**OIE**	**FAO**	**CBD**	**IMO**	**ATS**	**NAFTA**	**APEC**	**SPREP**	**BC**	**AU**	**NZ**	**US**
Recreation		●									●			
Bird watching													❍	
Fishing									❍					
Option value		●												
Existence value		●					●							
Bequest value		●									●			
HUMAN HEALTH	■					■			●		●	■	●	●
Human welfare									●					
Lost income									●		●			
Toxicity													●	
Human pathogen or parasite														●
Allergen levels													●	
Morbidity													●	
Mortality									●				●	

Discussion

This review has analysed the biosecurity risk assessment frameworks for international, regional and national instruments, with the goal to improve understanding and effectiveness of aquatic biosecurity efforts. Specifically, this review aimed to determine the number and scope of aquatic biosecurity risk assessment frameworks within the instruments; and to determine and compare the associated principles (e.g., use of precaution and uncertainty, the components of each risk assessment and the values used in each framework's consequence assessment) for the national, regional, and international risk assessment frameworks. Within this latter objective, two notable ideas emerged – the dichotomy within the application of precaution and the role of perception when assigning risk. The conclusions and recommendations derived from this review suggest actions that, if implemented, will improve ANS management and reduce the risk of ANS introductions.

Number and scope of aquatic biosecurity risk assessment frameworks

This review found 6 of the 14 instruments had risk assessments specific to aquatic biosecurity, though none of these addressed biofouling. Of these, three were international (OIE, FAO, IMO), two were regional (NAFTA, BC) and one was national (US). The remaining instruments either had general biosecurity risk assessment frameworks (IPPC, ATS, AU, NZ), provided risk assessment guidance but referred to other instruments for development of the framework (WTO, CBD), or are in the process of developing a framework (APEC, SPREP).

Despite the availability of international and regional risk frameworks that are specific or can be adapted to ANS threats, the geographic scope of implementation is limited; few national instruments exist to deal with these hazards. While the tenant of "something is better than nothing" applies, this review emphasized the need for development and coordination of risk assessment frameworks. This could occur through increased lateral spread (e.g., Australia, New Zealand and United States providing guidance to Pacific Island Countries and Territories [PICTs]; McNeely 2000) or increased vertical implementation (van den Bergh et al. 2002) (e.g., IPPC providing assistance to individual Member States). The increase in the number of risk assessments adopted and implemented at the national level is not only necessary to reduce the threats under control of the particular State (e.g., vessels arriving to domestic ports), but also necessary to ensure actions to address the risk are not undermined by the absence of action in a neighbouring State. As the marine environment lacks physical borders, strict controls in one area may be rendered ineffective if a nearby area does nothing (van den Bergh et al. 2002).

The scope of the content within the risk assessment frameworks was somewhat limited, particularly for consequence assessment considerations. While the frameworks provided relatively extensive guidance for factors effecting and determining the likelihood component and examples of values to consider in the consequence assessment, few instruments provided considerations to determine or estimate consequence, particularly when there is a lack of existing information for the threat. This may not be the fault of the instruments, however, but result from a lack of knowledge on what considerations can be used to estimate consequence in the absence of sufficient information. This further supports the need for additional research not only on impacts of specific species, but also what factors and characteristics affect impact severity and frequency, and have potential for serving as a proxy for impact when specific data is unavailable.

Comparison between national, regional and international risk assessment frameworks

The principles in the national, regional and international instruments were fairly similar, with science-based assessments being the most common at each level. International cooperation was proportionally higher at the regional level than at the other two levels (Figure 2.1), which may indicate a role for regional instruments to provide guidance to developing instruments at the national level. The acceptable types of information were also similar at each level and reflected the makeup of the principles (with scientific literature the most common information type at all levels).

The distribution of risk frameworks specific to aquatic nonindigenous species was approximately equal between levels. The trend at the national level may be slightly misleading, however, due to the differences in the mechanism of risk assessment. The US provides a risk assessment process that is aquatic nonindigenous species-specific. In contrast, Australia and New Zealand provide a broad description and methods for the risk assessment, which can be adapted to detailed Import Health Standards for individual hazards (e.g., ornamental fish). Thus, while their standard risk assessment process is not specific to aquatic biosecurity (and hence, does not contain invasion biology terminology; see below), they have the capacity to complete risk assessments more focused on aquatic biosecurity as the need arises.

The terminology regarding the species under consideration depended less on the level than on whether the instrument had a risk framework specific to aquatic biosecurity. Those instruments without a framework specific to aquatic biosecurity tended to use general terminology (e.g., ‘disease’, ‘pest,’ or ‘pathogen’). Conversely, those instruments with frameworks specific to aquatic biosecurity tended to use terminology specific to invasion biology - albeit a variety of terminology that lacked consistency between levels. One exception to this trend was the OIE, which contained general terminology despite having a framework specific to aquatic biosecurity. This may be explained by a combination of the history and purpose of the OIE *Aquatic Animal Health Code*. The *Aquatic Animal Health Code* was first published in 1995 after the WTO named the OIE as a standard setting body for animal diseases (OIE 2009b), which explains the similarity in terminology. However, while the OIE risk assessment technically applies to aquatic biosecurity, the purpose is different from other aquatic biosecurity-specific frameworks in that the OIE focuses on the diagnosis and prevention of diseases in aquatic animals of commercial value. Thus, the OIE regulates ANS not for the sake of environmental protection, but for protection of commodities (animals) valuable for trade purposes.

Two exceptions of the opposite nature were the CBD and ATS. Both had terminology specific to invasion biology but no risk framework specific to aquatic biosecurity. The CBD as an exception may be due to the fact that although it contains sections specific to the management of nonindigenous species for the purpose of protecting environmental values (i.e., biodiversity), it is not a standard setting body and only provides guidance for an aquatic biosecurity risk assessment framework. The ATS Madrid Protocol, despite containing some of the same signatories as the WTO and having been created around the same time, has a different purpose (i.e., protection of the Antarctic environment, as opposed to protection of trade). Therefore, it is unsurprising that the terminology is more similar to instruments such as the CBD. That it does not contain a risk framework specific to aquatic biosecurity may be due to the general nature and early stages of the risk assessment guidelines; while the prohibition of species introductions applies to water, there is no specific aquatic risk assessment framework (British Antarctic Survey 2007). The general inconsistency between terminology in all instruments supports previous studies and calls to harmonize aquatic biosecurity vocabulary (Subsidiary Body on Scientific Technical and Technological Advice 2001, Occhipinti-Ambrogi and Galil 2004).

The likelihood components showed no differentiation by level. However, at the international level, the inconsistency between the WTO and the IPPC/OIE was notable. As the IPPC and OIE are standard setting bodies for the WTO, it might be expected that the general approach (including the likelihood assessment) would be similar. The WTO framework, however, defined the likelihood as the probability of “entry, establishment *or* spread” while the IPPC framework defined the likelihood as the probability of “entry, establishment *and* spread” and the OIE as the “release *and* exposure” (WTO 1995, IPPC Secretariat 2004, OIE 2009a). Thus, there is a difference in the process of estimating likelihood (‘and’ versus ‘or’) based on the defined endpoints.

The presence of core values for consequence assessment was similar among levels, although the international and national instruments had a slightly higher representation rate of core values, which was likely due to the absence of specific framework for two of these instruments at the regional level (APEC and SPREP). The ATS was notable as the only organization that provided examples of consequence values that didn't include any in the economic category.

The application of precaution varied somewhat between levels. Approximately half (57%) the international instruments and all the regional and national instruments included precaution in some form (explicitly or non-explicitly) (Subsidiary Body on Scientific Technical and Technological Advice 2005, British Antarctic Survey 2007, IMO 2007, Arthur et al. 2009). However, where it was included, a definition was often provided but little else. There was a general lack of guidance on how precaution could be implemented into the risk assessment or aquatic biosecurity policies. Where it was suggested, it was generally in the form of 'clean' lists (e.g., New Zealand). In this approach, a hazard is assumed to pose an unacceptable risk until a risk assessment proves otherwise. Discussion of precaution is continued in "Duelling priorities for precaution: economy and environment" (next section).

The documentation of uncertainty was required at each level in approximately the same proportions (71% internationally; 50% regionally; 66% nationally). Of those instruments that called for recognition of uncertainty, several included a discussion of types of uncertainty. These tended to fall into two types: uncertainty surrounding the biological information, and uncertainty surrounding the assessor/assessment framework. Uncertainty surrounding the biological information was generally attributed to a lack of such information and as such, the instruments called on additional research as a solution. The NAFTA and US frameworks provided an example of how to partially manage this type of uncertainty by providing the assessors an opportunity to rate the uncertainty surrounding the individual components of the assessment (this rating also includes uncertainty of the assessor; e.g., NAFTA). Uncertainty surrounding the assessor/assessment framework includes uncertainty on the part of the individual (e.g., variability between assessors may lead to a different assessment for the same set of data), and uncertainty as to the framework components (e.g., which consequence core values and subcomponents are important to include). These two sources were addressed using two different methods: the NAFTA and US method of an uncertainty rating, and the FAO and BC method of using a Delphic process. The Delphic process identities and addresses this form of uncertainty using group discussion; the results of group opinion are repeatedly taken back to the group until agreement is reached on components of a risk framework (e.g., core values). Individual uncertainty can be identified via the variability of opinion within the group. Several of the instruments (WTO, FAO, NAFTA, BC, SPREP) identified the role of (risk) perception in risk assessment and its influence on this latter type of uncertainty, which is further discussed below, in "Perception of risk assessment".

Duelling priorities for precaution: economy and environment

Recent decades have seen the use of precaution in measures and actions that attempt to anticipate a threat's adverse impacts to the environment, in the absence of scientific certainty (Goldstein 2001, Peel 2005, Peterson 2006). The most widely cited definition is Principle 15 of the UN Rio Declaration on Environment and Development (United Nations General Assembly 1992), which states that in case of "threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation".

Of the agreements reviewed, the CBD, IMO, APEC, SPREP, BC, Australia and New Zealand explicitly include the use of precaution when performing a risk assessment. For the remainder (WTO, IPPC, OIE, ATS, NAFTA and the US) its application is unclear and left up to interpretation (e.g., via provisional measures). Most significant of these is the WTO, as it establishes the risk assessment principles implemented via other instruments (e.g., the OIE and IPPC), and is one of the highest authorities of international agreements (Campbell et al. 2009).

In the WTO SPS Agreement, precaution is not specifically mentioned, but may be relevant through reference to extra-WTO sources (e.g., the CBD) (Cheyne 2007). Article 31(3)(c) of the Vienna Convention of the Law of Treaties mandates that any relevant rules of the international law applicable in the relations between the parties shall be 'taken into account' in treaty interpretation (McLachlan 2005). Thus, if precaution is accepted as international law, it may be permissible for countries to use it within the WTO context. Whether precaution has indeed reached the status of international law is uncertain (Birnie and Boyle 2002). It is accepted as law at regional (within the European Community) and, some would argue, international levels (Kogan 2006). Within the WTO, however, decisions by the WTO's two dispute boards (Dispute Settlement Panel and the Appellate Body) have remained unclear or negative in their views on precaution as international law. Prohibiting the use of precaution places the burden of proof on the proponent of the measure to provide supporting scientific evidence for the measure, which is often impossible due to the lack of sufficient evidence. This makes it difficult to develop measures to prevent the introduction of nonindigenous species, especially ANS, where information necessary for consequence assessment is scarce.

Although interpretation of the SPS Agreement that allows the use of precaution for environmental measures would facilitate the reduction of the ANS threat, there is another, largely unrecognized application of precaution within SPS standards for measures and risk assessments that may still be more influential. That is, precaution is generally discussed in the context of environmental measures, but the widespread use of precaution in the context of trade, and specifically, the protection of trade, is largely overlooked (Campbell et al. 2009). An example of this is the use of dirty lists (e.g., the WTO and US approach), as opposed to 'clean lists' (e.g., the CBD or New Zealand approach). 'Dirty lists' allow imports of all organisms unless specifically regulated (e.g., due to a finding of high risk). These lists therefore assume an "innocent until proven guilty" (i.e., non-precautionary) status for imports (Takahashi 2006). Due to this approach, critics of the SPS Agreement claim it is insufficient to keep out introductions, as it prioritizes protection of free trade over protection from nonindigenous species (Campbell 2001).

Conversely, 'clean lists' prohibit imports of any organism not specifically approved (e.g., risk level is within the acceptable level of risk). These lists therefore assume a "guilty until proven innocent" (i.e., precautionary) status for imports (Takahashi 2006). The CBD and New Zealand applies this approach to nonindigenous species, and places the responsibility for proving innocence on the exporter (Campbell et al. 2009). Although the CBD is a generally accepted international agreement (ratified by Australia, New Zealand, Canada and the European Union, though not the United States), the WTO is still the overriding treaty for measures that could impact trade (Shaw and Schwartz 2005). Thus, even if precaution in establishing environmental measures has a role in the WTO agreements, the "innocent until proven guilty" attitude toward imports remains and the impetus is on the importer to perform the risk assessment and defend the measure. The WTO will still favour precautionary protection of trade over precautionary protection of the environment (Hewitt and Campbell 2007).

Perception in risk assessment

The review also found that several frameworks include the concept of risk perception in identifying threats and consequences. In the WTO *EC – Hormones* dispute, the Panel asserted that non-scientific considerations (such as cultural or moral preferences and societal value judgments) may not be included in a risk assessment (Pauwelyn 1999). The AB disagreed, however, stating that consideration was not limited to those factors explicitly listed in the SPS Agreement and that "the risk that is to be evaluated in a risk assessment under Article 5.1 is not only risk ascertainable in a science laboratory operating under strictly controlled conditions, but also risk in human societies as they actually exist...” (Pauwelyn 1999). In the WTO *EC-Asbestos* dispute, the AB rejected the Panel’s assertion that the two products (asbestos and imitation fibre) were “like”, indicating the difference in risk to human health between the two products indeed made them “unlike”; not only should physical properties of the product be considered, but consumer perception and behaviour (i.e. risk perception) as well (Goldstein and Carruth 2004).

The FAO suggested that uncertainty in economic or socio-political knowledge and information is due to variations in perceptions of consequence between cultural groups (Arthur et al. 2009). The BC SPA/BD Protocol Guide states that both the real and perceived consequences of a species should be examined against the core values of the region, and the perceived consequences can be assessed using normative evaluation (in which the degree of consequence is determined by a value judgment of the difference between impacted and non-impacted value). Finally, the US ANSTF Risk Framework and NAFTA CEC Guidelines recognize scientifically-defensible, anecdotal, and experience-based information and acknowledge that all types of information will be affected by human perception.

These sentiments reflect the current trend in risk management (National Research Council 1994). Risk was originally viewed as something that existed externally, governed by physical processes that could be measured in a quantitative fashion (Slovic 1999). Scientists and risk experts were seen as rational, objective assessors of risk (DuPont 1980, Weiner 1993). These views have changed, however, due in part to the growing appreciation of the subjective nature of risk (Slovic 1999). Current risk theory rejects the concept of risk as purely objective, instead describing risk as a subjective concept created by humans to help deal with the hazards and uncertainties in life (Krimsky and Golding 1992, Pidgeon et al. 1992). This subjective view of risk is now generally accepted in most risk literature and by many risk-management agencies (National Research Council 1994, 1996a).

Challenges to and weaknesses in aquatic biosecurity risk assessment

Several challenges and weaknesses associated with the aquatic biosecurity risk assessment frameworks were identified in the review, including a lack of national implementation of the international and regional frameworks, scarcity of ANS impact data, little guidance on consequence assessment methodology, limited number and scope of standards and measures related to ANS, and a variety of terminology between frameworks. These are discussed below.

Although this review found that six of the 14 instruments provided frameworks for aquatic biosecurity risk assessments and five of the six were binding (i.e., enforceable ‘hard law), this has not translated into the widespread development of aquatic biosecurity risk assessment policy or practice. This weakness may be due to the need for national governments to implement the standards in order for them to be effective; the international standards themselves are not legally binding, but must be enforced by the countries themselves (IPPC Secretariat 2002). This is often an expensive and lengthy process, generally leading to less than 100% participation by the Members (Shine et al. 2000). Even when Members are interested in complying with a standard or agreement, they face difficulties in understanding how to apply the objectives (e.g., CBD) or lack the capacity or resources in implementing the objectives (Campbell et al. 2009).

The most common principle in the reviewed frameworks was a science-based risk assessment, reflecting the increasing role of science in international trade agreements (Wirth 1994). Yet, at the same time, many of the frameworks recognize the challenge presented by the paucity of available information, particularly for consequence assessment. For example, the IMO G7 Guidelines acknowledge the difficulties presented by lack of data. The FAO framework highlights that uncertainty is associated with poor or incomplete biological knowledge (e.g., how an organism will react to specific stimulus and what impact an organism will have on another organism). The OIE framework states that a qualitative risk assessment is often sufficient, in light of the paucity of data, particularly for aquatic species introductions, and the CBD decisions find a scarcity of consistent and comprehensive data, making risk assessment difficult. The requirement for scientific data juxtaposed against the general lack of such data reinforces the need for additional research on ANS and their impacts.

In addition to the lack of data, there was a lack of guidance for determining consequence in the frameworks. Although many of the risk assessment frameworks provided consequence core values and subcomponents, these provide little help if data is lacking; it is often necessary to make decisions with little or no impact data. The frameworks did not account for these situations in which the assessor is lacking impact information but is required to make a decision on the consequence and risk posed by the threat using other considerations (e.g., characteristics of the species). Even when evidence is sufficient, there was little guidance on criteria for measuring consequence or determining the threshold values that differentiate levels of consequence (as used in semi-quantitative and qualitative assessments).

Another weakness is the limited number and scope of standards and measures (including risk assessment) established by the IPPC, OIE and other international instruments relevant to environmental and biosecurity protection. As of 2010, there are few trade-related international standards, especially those that address specific organisms or pathways – and none that deal with aquatic organisms (Shine et al. 2000, Hedley 2004). The IPPC has three related to pest risk analysis, but these only address plant or plant pests, not unintentional introductions of non-plant pests that may arrive with plants (Shine 2007). Another gap in standards, discussed in several CBD COP Decisions, is the absence of measures relating to animal pests that are not specific pests of plants or animals (particularly nonindigenous freshwater species) (Subsidiary Body on Scientific Technical and Technological Advice 2001). So although provisions in the IPPC and OIE standards for indirect effects of pest species may provide for regulation of some nonindigenous species (e.g., through their effects on ecosystems, which can indirectly effect plants), a nonindigenous species may not be regulated under IPPC or OIE standards unless it is specifically added to the list of plant pests (IPPC) or international notifiable diseases (OIE) (Subsidiary Body on Scientific Technical and Technological Advice 2001).

An additional weakness is the variety of language between risk assessment frameworks discussing these pests and/or nonindigenous species. Although many of the instruments with risk assessments general to these organisms use similar language (i.e., pest or pathogen), aquatic biosecurity risk assessments (those by the FAO, IMO, NAFTA, APEC and SPREP and the US) included a variety of terms such as 'invasive', 'exotic', 'non-native', 'alien', 'harmful', and 'nonindigenous'. This supports the finding by the CBD SBSTTA that inconsistent or undefined terminology makes interpreting the purpose and intent of the text difficult (Subsidiary Body on Scientific Technical and Technological Advice 2001). As in this review, the SBSTTA found that some agreements refer to species that are not endemic and may or may not pose a threat (e.g., 'alien', 'nonindigenous', 'non-native', 'exotic', 'foreign', 'new', or 'introduced') and some refer to species that are not endemic and do pose a threat (e.g., 'invasive', 'weed', or 'pest'). Even with the same intent (e.g., indicating a non-endemic species that doesn't pose a threat), terms differ in the attitudes they evoke. For example, 'introduced' is relatively neutral, while 'alien' is somewhat negative, and 'nonindigenous' is somewhere in the middle (Shine et al. 2000).

The issues identified in this review support additional findings in the SBSTTA review, which analysed 39 binding and non-binding international instruments on alien species. Within these instruments, the SBSTTA found a: lack of national systems implementing the international agreements; paucity of consistent and comprehensive data, making risk assessment difficult; and conflicting or undefined terminology within and between instruments (Subsidiary Body on Scientific Technical and Technological Advice 2001). The scarcity of national legislation by signatories to international instruments related to nonindigenous species is found in other studies, as well (van den Bergh et al. 2002, McGeoch et al. 2010). McGeoch and colleagues (2010) found that only 55% of CBD signatories have relevant national legislation, and of this legislation, much is poorly implemented. In addition, the review found that multilateral environmental agreements lacked details on treaty obligations (e.g., CBD), sanitary and phytosanitary instruments lacked biodiversity criteria in risk assessment frameworks (e.g., IPPC and OIE), technical guidance in the transport sector was nonbinding and lacked coverage over all pathways and vectors (e.g., vessel biofouling under the IMO (Roberts and Tsamenyi 2008)), and instruments to regulate intentional introductions lacked binding instruments for aquatic introductions (57% instruments reviewed) (Subsidiary Body on Scientific Technical and Technological Advice 2001).

Recommendations for aquatic biosecurity risk assessment and policy

The outcome of this review suggests several steps to improve biosecurity risk assessment.

- **Due to the necessity of implementing national guidelines (i.e., international standards requiring implementation at the national level) and associated difficulties (e.g., the financial and technical resources available versus the resources required), guidance for implementing international standards should be provided through lateral or vertical means. The need for assistance in developing straightforward risk assessment frameworks is particularly strong for developing countries or countries that lack sufficient technical resources but often include valuable aquatic habitat.** This could occur via two methods: lateral spread of existing frameworks or vertical integration of international or regional standards. Lateral spread could occur via countries with (aquatic) biosecurity risk assessment frameworks (i.e., Australia, New Zealand, the United States) sharing their methodology and assisting other nations to adapt it to their specific requirements and conditions. Vertical integration could occur via increased assistance of international and regional organizations to individual nations. This approach is supported by the review, which noted “international cooperation” as a common principle, especially at the regional level. Regional assistance may be the most practical, as regional organizations would be the most likely to have knowledge of the characteristics of the region (e.g., vessel traffic or species compositions) and differences (by nation) within the region. Increasing the number of nations with risk assessment frameworks would both promote aquatic biosecurity and also fulfil the risk assessment requirements under the WTO SPS Agreement.

- **Additional research should focus on ANS impacts and development of consequence assessment methodology for all core values to improve the consistency and transparency within and between risk assessments. For example, this would include considerations for determining consequence in data-scarce situations and guidance on criteria for measuring**

consequence and determining threshold values to differentiate levels of consequence. This review supports the views of Parker and colleagues (Parker et al. 1999) in finding a lack of research and development for consequence assessment considerations. Particularly, research should focus on developing considerations that can be used to estimate the potential consequence of a species, and providing guidance on criteria that can be used to measure consequence and set the threshold values that differentiate levels of consequence. Most effort has focused on the economic component, although this should diversify to include environment, social, human health and/or cultural components (as supported by the repeated inclusion of each). The majority (66%) of instruments that specified consequence direction focused on negative consequences, which suggests that the limited resources available should be used to address the negative consequences before attempting to incorporate any positive aspects of ANS introductions. In addition to finding a lack of research on consequence assessment frameworks, this review found five of the instruments (FAO, IMO, NAFTA, BC, US) highlighted the lack of research on impacts of individual ANS and cited it as a significant source of (biological) uncertainty. Given limited funding, additional research could be prioritized according to assessed uncertainty and potential consequence. That is, species with high uncertainty and low consequence may actually have serious impacts that should be investigated before assumed to represent low risk (see Chapter 4, Figure 4.8 for additional discussion).

Of the two research priorities, the most productive area for further information gathering from a biosecurity perspective includes factors effecting the categorical descriptions of impact, e.g., effect size/thresholds and acceptable rates of Type II errors (see Figure 5.3). This would include a focus on processes that set threshold levels and on facilitating comparison of acceptable rates of Type II errors and the costs of Type II errors, respectively. Understanding these factors is important given the nature of biosecurity risk assessment management. Biosecurity agencies often base decisions on policy-driven entities such as the Acceptable Level of Risk (ALOR), as well as economic limitations, and a better understanding of these fundamental factors would facilitate more effective decisions. Although, species-specific impacts are also important, there are simply too many species for a comprehensive understanding of each within any realistic timeframe or budget. Establishing a framework for processing these species using available knowledge would be more effective from a biosecurity perspective.

- **Organizations such as the IPPC and OIE should work to increase the number and scope of standards and measures (including risk assessment) specific to ANS and associated vectors.** The need for internationally accepted standards is evidenced by the findings of this review. For example, there were only six aquatic biosecurity risk assessment frameworks and a lack of a risk assessment framework to address vessel fouling. This situation can be ameliorated in two ways. First, the absence of ANS standards and measures could be addressed by the IPPC and OIE increasing the coverage of organisms under their standards (i.e., regulating plants and animals, respectively that cause harm to any of the core values, not just to other plants and animals). Second, the absence of vector-specific frameworks can be addressed by taking advantage of the multidirectional interaction between international and national systems. That is, national governments could influence the adoption of standards through recommendations and input to international instruments (Hewitt and Campbell 2007). This has already been initiated for vessel fouling, with New Zealand providing leadership by adding vessel fouling to the IMO agenda.

- **National, regional and international instruments should develop a shared set of ANS-specific terminology to reduce discrepancies and provide for harmonization between risk assessment frameworks.** The variety of terminology (Table 2.4) is not only confusing it can incorrectly describe the status of a species. For example, "invasive" is a common term often used synonymously with a nonindigenous pest species (or, even just a nonindigenous species). However, native species can be invasive and not all introduced species are invasive (i.e., have impacts). Invasive may be appropriate in certain circumstances but care should be taken to use the term appropriately. Nonindigenous is an innocuous term that may be more appropriate for widespread use. Another consideration is the switch between "marine" and "aquatic". The use of "marine" may be too limiting; "aquatic" indicates that the assessment includes not only marine species, but also those in estuarine and freshwater systems. While it may be impractical and unnecessary to retroactively harmonize terminology within existing documents, developing a shared set of ANS vocabulary will ensure future risk assessment frameworks will be clear and accurate.

- **The use of "clean lists" should occur where possible to provide a standardized method to incorporate precaution into aquatic biosecurity policy while still fulfilling the obligations under the WTO.** Precaution was identified as an important consideration in many of the instruments (eight specifically included it and three may allow for it). However, the instruments that mentioned precaution failed to provide guidance as to how it could be incorporated into the risk assessment and risk management process. An approach taken by several of the instruments that indirectly but consistently incorporates precaution is the use of “clean lists”. These lists assume a “guilty until proven innocent” (i.e., precautionary) status for imports by prohibiting imports of any organism unless a risk assessment has shown the risk level to be within the acceptable level (Takahashi 2006).

- **Given the increased recognition of perception in the field of risk and risk management, additional research should examine the role of perception in aquatic biosecurity and risk assessment to improve the understanding and facilitate the integration of perception, subjective judgment and the associated assumptions into risk assessment frameworks.** The potential role of judgment and perception to aquatic biosecurity risk assessment is highlighted by the fact that none of the instruments mandate quantitative risk assessments, allowing for potential adjustments or other considerations related to the effects of perception (e.g., differences in the thresholds for consequence categories). In addition, many of the instruments allowed the use of a variety of information types that inherently involve judgment and perception (e.g., stakeholder, experience-based and expert judgment). The use of semi-quantitative and qualitative risk assessments, combined with the need for a transparent process (a principle in many of the instruments), calls for a methodology that includes the variety of information types in a scientifically-defensible and comprehensive manner. While a full proposal of this methodology is beyond the scope of this chapter, it would potentially involve the Delphic process, as described by the BC and FAO, combined with additional research on the factors effecting biases and perceptions by experts in the ANS field.

The following chapters address several of the gaps identified in the review and solutions described in the recommendations. They dissect the solutions into specific actions or research needs. However, there are many more critical questions not identified in this chapter highlighting additional data gaps that effect biosecurity risk assessment. How will climate change affect the risk to core values? Specifically, how will this change affect the transport (natural and anthropogenic) of ANS? Will the stress on ecosystems increase invasibility? How do we address cryptogenesis, particularly in rare species? How do expanding free trade agreements affect the risk of ANS? Answering these questions remains outside the scope of this thesis, but would constiute a valuable exercise.

CHAPTER 3. MITIGATING UNCERTAINTY USING ALTERNATIVE INFORMATION SOURCES AND A MODIFIED DELPHIC PROCESS IN AQUATIC NUISANCE SPECIES CONSEQUENCE ASSESSMENT

Manuscript in review:

Dahlstrom, A., C. L. Hewitt, and M. L. Campbell. 2011. Mitigating uncertainty using alternative information sources and expert judgment in aquatic nonindigenous species consequence assessment. Aquatic Invasions (in review).

Dahlstrom, A (60%), Hewitt, CL (20%), Campbell, ML (20%)

Introduction

Evaluating the consequence of an adverse event as a component of risk assessment can occur via direct assessment of empirical data, or heuristic assessment by experts and/or stakeholders that is either used directly or assessed by a trained risk assessor (Hewitt et al. 2010). In particular, heuristic assessment uses expert and/or stakeholder judgment and perception (based on personal knowledge and available information) to mitigate uncertainties and make decisions (Scheffran 2006). Heuristic assessments can include one or more assessors in an independent or group setting. The latter often occurs via a Delphic process, where it is developed for making decisions and predictions in conditions of scarce and/or highly uncertain information inappropriate for traditional scientific methods. This process occurs in a variety of fields, including a range of areas within environmental decision-making, from conservation of threatened species (Marsh et al. 2007), to application of sewage sludge to farmland (Webler et al. 1991 and assessing drinking water contamination (Shatkin and Qian 2004). The conventional Delphic process generally includes defining the problem, selecting experts, presenting background to experts, obtaining expert assessments, aggregating and re-distributing the results to experts, and repeating these steps until consensus is achieved (Linstone and Turoff 2002, Burgman 2005). While sometimes considered demanding on time and other resources by the experts involved (Webler et al. 1991), real-time, single-iteration group exercises are time- and cost-effective for activities such as risk assessment that may otherwise require months or years to complete using other methods (Krueger and Casey 1994).

In an aquatic biosecurity context, the Delphic process is useful in several ways. When assessing consequence, it facilitates a decision-making process based on knowledge from a variety of experts and/or stakeholders (including that from scientific and technical experts, government agencies, fishery managers, stakeholder/community groups, industry, recreational or conservation organizations and/or indigenous groups). While scientific or other evidence can aid the process, it is not required (e.g., Malchoff et al. 2005). In addition, the process can guide development of the consequence assessment framework and methodology, such as choosing and defining the core values and categorical thresholds (e.g., Campbell 2008). Thus, it provides an opportunity to include a variety of individuals representing all core values, which can lead to increased acceptance of the risk assessment outcomes and facilitate the process of risk communication (Beale et al. 2008) and risk management (Marsh et al. 2007).

The traditional Delphic process has several limitations, however. For example, it does not deal with uncertainty, discourages differences of opinion, and potentially leads to incorrect interpretation and aggregation of results by the facilitator (Linstone and Turoff 2002, Burgman 2005). As such, several variations on this traditional method have been developed (e.g., lack of anonymity or absence of final consensus). In a modified group Delphic process, participants meet in a group setting where discussion of uncertainty occurs, followed by individual assessments, a reconvening of the group to discuss differences, and finalized by a presentation of the results that retains the content of each assessment (i.e., does not force consensus) (Webler et al. 1991). Webler's et al. (1991) study explored several of these modifications (using direct discussion, no consensus requirement, and analysis of uncertainty), and found them effective for addressing weaknesses, while still preserving the process's key elements: the defining feature of the Delphic process is allowing expert revision of judgment based on input and opinion of other experts to reach consensus where possible and identify areas of disagreement where consensus is not possible, with a subsequent reduction in overall uncertainty (Webler et al. 1991).

Uncertainty in consequence assessment

Due to the predictive nature of risk, uncertainty is an inherent component of risk assessment (Morgan and Henrion 1990, Walker 1990, Pollack 2003). Knight (1921) used the term uncertainty to describe events to which exact probabilities cannot be applied (e.g., the chance of a nonindigenous species escaping from a local aquarium in twenty years), whereas risk events are those with unknown outcomes but known probabilities (e.g., the chance of a species with a consistent reproductive cycle releasing viable offspring during a vessel's port visit). This uncertainty can stem from a variety of factors, including: (1) knowledge gaps; (2) systematic and random measurement error (flawed measurements and uncertain or inappropriate models); and (3) variability (the variety of impacts that the same hazard can have in different locations of time and space) (Klinke and Renn 2002). In addition, ambiguity leads to increases in each of these types of uncertainty (Klinke and Renn 2002). Ambiguity results in uncertainty due to different norms and values leading to, for example, different interpretations of the same data set. In addition to remaining an essential element of the risk assessment process, understanding the sources of uncertainty provides advantages to the policy application of the assessment, including identification of potential policy focus areas within the risk issue, an increase in the transparency of the assessment outcomes and easier revision or adaptation of these estimates for other use (Morgan and Henrion 1990).

<u>Knowledge gaps</u>: As discussed in Chapter 1, ANS knowledge pertaining to core values often contains significant gaps related to the likelihood, and particularly the impacts, of the introduction, spread, or establishment of a species (Grosholz and Ruiz 1996, Parker et al. 1999, Lovell and Stone 2005, McGeoch et al. 2010). For example, the significant knowledge gaps in the taxonomy and

biogeography of many species (Hewitt et al. 2004a) may result in the (mis)classification of a nonindigenous species as indigenous, assignation of different names to the same species in their native and introduced range, or uncertainty as to whether a species is indigenous or not (i.e., cryptogenesis; sensu Carlton 1996a). Chapman and Carlton (1991) conclude that these errors have resulted in an underestimation of nonindigenous species numbers (which may lead to an underestimation of risk). This difficulty is further confounded when cryptogenic species resemble threatened, endangered or protected species. Assigning a risk to a species that may occur on two opposite ends of the “desirability” spectrum is an uncertain process with significant implications for risk assessment decisions.

Systematic and random measurement error: Systematic and random measurement errors result from biases or errors in collecting or interpreting measurements. In an impact assessment context, this may stem from the methods and assumptions of Null Hypothesis Significance Testing (NHST). Scientists traditionally use NHST to test hypotheses, which is based on comparison of the *p*-value to a chosen significance level (generally α), below which the results are considered statistically significant; the two “treatments” (generally one treated and one untreated control) are assumed the same unless the analysis indicates otherwise. For impact assessment, particularly where the impact is potentially catastrophic and unmanageable, the implications of this approach are significant. The assumption that treatments are the same (i.e., generally the null hypothesis states the treatment will have “no effect”) results in higher rates of Type II errors (missing an impact) versus Type I errors (falsely assigning an impact). This problem is exacerbated by low power, which is common in ANS impact studies with small sample size (i.e., even if the impact is significant, the analysis will fail to find significance due to inadequate experimental design). Chapter 5 provides a further discussion and exploration of this concept. In addition, given the often unpredictable and unexpected nature of species introductions, adequate experimental designs, such as before-after-control-impact (BACI), are often impossible. Given the rarity of existing and appropriate pre-impact data, a researcher would have to guess at where and how the impact would occur in order to obtain necessary data – which is often considered an impractical task.

Within ecological research, impact assessment has been described as perhaps the least appropriate area for the traditional statistical focus on avoiding Type I errors. Several authors, such as Page (1978), Toft and Shea (1983), Peterman (1990a), and Fairweather (1991), have suggested that in some situations (e.g., natural resource management) or over the long-term, Type II errors will be more costly than Type I errors. This means that Type II errors will not only incur the environmental costs when management fails to take early action to prevent or minimize the impact, but they will also incur economic and/or regulatory costs once the impact is realized – at which point the damage to the resource or ecosystem service may be irreversible. Thus, the consistent use of NHST indicates a willingness to accept Type II errors (missing an impact) rather than Type I errors (falsely assigning an impact) and may result in a bias against environmental conservation and sustainable management (Mapstone 1995).

Ambiguity: Ambiguity results from different interpretations of the same set of conditions due to conflicting norms, values or definitions. Ambiguity can affect the basic components of risk (e.g., how risk is defined; Cox 2008), as well as the more specific components of risk assessment (e.g., how to measure impact) (Burgman 2005). In the aquatic biosecurity context, ambiguity may occur while developing the risk assessment framework (e.g., choosing the focus of the risk assessment, the values to include and appropriate thresholds) or while performing the risk assessment (e.g., assigning consequence levels) due to different perceptions by experts and stakeholders (Campbell 2008).

Uncertainty in risk management

Those who apply the risk assessment outcomes (e.g., natural resource managers and policy makers) often view uncertainty as undesirable and call for an unambiguous risk estimate, leading to potential conflict with those involved in assessing the risk (e.g., scientists and stakeholders) (Slovic et al. 1979). In this way, uncertainty or insufficient proof surrounding a threat has (intentionally or unintentionally) led to delayed or absence of regulation or other policy decisions (Andrews 2003, Pollack 2003). As such, efforts to address uncertainty are an essential element central to the risk assessment process itself that can not only improve the risk assessment outcomes but also lead to greater overall understanding of the risk and, when uncertainty exists, guide research and management efforts (Pollack 2003).

Risk assessors often look to reduce uncertainty via technical means (Peel 2005). However, attempting to merely reduce uncertainty via additional research, for example, may not be practical due to limited financial and personnel resources, or possible given the variety of uncertainty origins (e.g., high variability, the influence of social or cultural considerations, or the role of subjective judgments; Westman 1985, Stewart 2000). While expert judgment is often the most appropriate method to make consequence and risk estimates under conditions of uncertainty, and particularly data scarcity (Halpern et al. 2007), experts often prefer using empirical data to make decisions. When knowledge gaps force experts to rely on other means to make decisions, experts have several options, including using alternative information sources, or (when lacking alternative information sources) precaution.

Alternative information types: Despite repeated calls for more empirical work for ANS, there still exists a lack of empirical data, particularly for certain core values or taxa, such as social and cultural values or freshwater species, respectively (Gherardi 2007). Even in areas that traditionally receive significant attention, (e.g., economic impacts) many of the studies (where they exist) are anecdotal in nature (e.g., de Oliveira et al. 2006).

Ecological assessments may also rely on anecdotal information (Parker et al. 1999); and, given the absence of any data, some advocate using sources other than empirical literature (e.g., local knowledge; Mackinson 2001). Some ecologists posit that a focus on manipulative experiments alone limits the range of questions an ecologists can explore (Hobbs et al. 2006). Others, however, believe that the impacts and behaviour of ANS can only be properly addressed through a large amount of quantitative information to allow the development of tools such as predictive modelling (Gherardi 2007). Perhaps most important in the policy context, however, is whether an assessment based on alternative information sources will satisfy WTO mandates (e.g., the SPS Article 5 mandate for a *science-based* risk assessment in order to apply any SPS measures that may restrict trade.) To this end, this study will gather expert opinions on the scientific validity and appropriate use of various information types (gathered from literature and discussion with colleagues; Appendix B: Table 1).

Precaution: In near or complete absence of data, ANS experts generally must take one of two contrasting approaches: assume a newly detected species is “guilty until proven innocent” (herein, the “precaution” approach) or assume a newly detected species is “innocent until proven guilty” (as suggested by the WTO; see Chapter 5, “Criticisms” for issues with this approach). Essentially, precaution allows action even without full scientific certainty in an effort to find the right mix of caution. The right mix is enough to avoid significant impact, particularly if the impact is irreversible, but not so much as to incur unreasonable costs (Klinke and Renn 2002), and includes using all available scientific knowledge to evaluate the risk and make appropriate decisions (Chowdhury and Sabhapandit 2007). Despite this philosophical stance, many policy decisions (such as the WTO SPS Agreement) have failed to explicitly include or allow precaution within environmental measures, often favouring the prevention of impacts to trade (Campbell 2001, Campbell et al. 2009). Even when the intent to use precaution is present, there are several challenges to its transparent and objective implementation, such as multiple definitions (Appendix B: Table 2; Tucker and Treweek 2005).

Where application of precaution is appropriate, there are numerous ways it can be incorporated: take action before scientific proof is available; use best available information; consult with stakeholders and interested parties; take measures to reduce uncertainty particularly when risks are high; to respond proportionally to the threat; and place the burden of proof on those proposing the activity (Peel 2005). These methods provide a foundation for the role of precaution in addressing uncertainty, and highlight the need to examine their potential application in other contexts, such as aquatic biosecurity.

This study examines the presence of and attitudes toward uncertainty and associated methods to address that uncertainty (i.e., modified Delphic process, alternative information sources, and precaution) in an aquatic biosecurity context. This includes an assessment of the effect of these methods on consequence assessment for ten ANS, using four different groups of ANS experts. This research was conducted in the United States (US) and Australia (AU) because both countries have fairly extensive biosecurity (including ANS) research and policy programs (allowing sufficient sample size). Both scientists and managers were included within the assessments, as these two groups were hypothesized to have different views on uncertainty and associated management solutions when presented with knowledge gaps, systematic measurement error, variability and ambiguity that prevent a direct, quantitative consequence assessment.

Methods

This study targeted both ANS scientists and managers. An 'ANS scientist' includes any individual involved in empirical research of ANS, with 'ANS manager' including any individual involved in decision or policy-making and management of ANS. Assessment was applied to each group separately in both the US and Australian workshops and involved:

1) an initial, internet-based survey;
2) a subsequent internet based survey; and
3) a consequence assessment.

Two sets of species were used in the survey and consequence assessment: five actual ANS and five hypothetical ANS (Table 3.1), with the latter mimicking actual ANS but described as "unknown." Ten species represented a compromise between achieving the objectives and keeping the time requirements within a feasible limit during assessment periods. The 10 test species were chosen to reflect a range of taxa, impact types, and distributions. For the five actual test species, the provided information was based on primary (peer-reviewed) and secondary literature (e.g., government reports and databases such as NIMPIS (http://adl.brs.gov.au/marinepests/) and NEMESIS (http://invasions.si.edu/nemesis/). For the five hypothetical test species, the provided information was generally modelled after one or a combination of actual species in the respective taxa. Impact information for each species was directed at one 'focus' core value.

Table 3.1. Ten ANS evaluated in the consequence assessment.

Species	Taxonomic affinity
Caulerpa scalpelliformis	Algae
Unknown Algae	Algae
Pterois volitans	Fish
Unknown Fish	Fish
Bonamia ostreae	Parasite/pathogen
Unknown Parasite	Parasite/pathogen
Maoricolpus roseus	Gastropod
Unknown Gastropod	Gastropod
Ciona intestinalis	Ascidian
Unknown Ascidian	Ascidian

The initial survey (Appendix B2) used 35 multiple choice and 3 open-ended questions to gather information on participant's demographics, background and worldview (questions 2-13); and attitudes toward uncertainty (14-20, 31), information types (as discussed in this chapter's Introduction; 32-34, 36-37), precaution (21-30, 35); and finally the preliminary series of ANS consequence assessments for the five hypothetical species (without provided information on the species or their impacts). This initial survey was followed by a second internet-based consequence assessment (Appendix B3) of all 10 species, which provided information on each species and their impact (in the form of a fact sheet and primary literature for four of the species). Both of these surveys were then followed by a final consequence assessment that involved group discussion (in addition to access to the earlier-provided information).

Consequence Assessment

The consequence assessment occurred in several stages to allow comparison of how assessment changed with additional data and group discussion. For each consequence assessment, participants rated impacts to each core value; the uncertainty associated with that assessment; justification for that assessment; any subcomponents for the four categorical values that participants felt were specifically impacted (specific elements within the broad core value; e.g., biodiversity for the environmental category); and the controllability, ability to mitigate, and general concern regarding the impacts of each test species.

The group discussion occurred via a face-to-face workshop that used a modified Delphic Process to determine how the assessment process worked for scientists and managers when assessing ANS consequences. The workshop consisted of a pre-discussion assessment, in which participants were asked as individuals to indicate their judgment of a species' consequences and the associated uncertainty in the four core values via a poster (one poster per group). This was followed by a participant group discussion that was concluded with a final post-discussion assessment (via same methods as pre-discussion assessment) and a final written survey (same format as second online survey, Appendix B3). When possible, the same participants completed all stages of assessment, although for logistical reasons, some participants were not able to either complete one of the surveys or attend the workshop (high drop-out rates are common in Delphic processes; Linstone and Turoff 2002). In these instances, data was retained for the analyses given the small sample size.

This study targeted a specific, often limited, group of participants, and hence non-random purposive sampling (Tongco 2007, Gillham 2008) that targeted experts in the ANS field was used. Given that participants needed to gather in one location for the workshop, as well as the limited funding and busy expert schedules, the workshops were held in conjunction with four conferences on ANS issues (Table 3.2).

Table 3.2. Conference name, location and date for each group workshop.

Group	Conference name	Location	Date
United States/Canada[7] (US/CA) scientists	6th International Conference on Marine Bioinvasions	Portland, OR, USA	24-27 August, 2009
Australian (AU) scientists	Australian Marine Sciences Association Annual Conference	Wollongong, NSW, Australia	4-8 July, 2010
US/CA managers	International Conference on Aquatic Invasive Species	San Diego, CA, USA	29 August-2 September, 2010
AU managers	Australian National Introduced Marine Pests Coordination Group meeting	Canberra, ACT, Australia	1-2 December, 2010

Ethics approval was sought and obtained from the Tasmania Social Sciences Human Research Ethics Committee, reference number H10726 (Appendix B4). At all times during this research the Australian National Statement on Ethical Conduct in Human Research was complied with. To maintain

[7] Although we targeted US participants, several participants were currently working or had trained in Canada. As such, we included Canada in the group description.

confidentiality, participants were provided with a participant number to use during the project, instead of personal name. While anonymity was lost during group discussion (as participants met face-to-face), analysis and reporting of results remained anonymous.

It should be noted that the final group discussion component limited the possible participant numbers for this study, given the difficulty of gathering experts together for such an exercise. As such, results should be generalized with caution. However, while the numbers of participants in each component were low enough to generally preclude statistical hypothesis testing, the participants represented a significant proportion of the sample population and hence these results should provide a representation sufficient for an introductory view into issues of uncertainty within consequence assessment.

Statistical analyses

Responses to survey questions, particularly those on uncertainty (general uncertainty, ambiguity, knowledge gaps, variability, and systematic and random measurement error), information type and precaution, as well as those from the two open-ended questions on challenges to consequence assessment, were summarized using descriptive statistics.

Several analyses were used to determine how the provided information and modified Delphic process affected the participants' assessments of consequence for the ten ANS, including summary and comparison at each of the following stages: (1) mean of each species' assessed consequences and standard errors for each core value at each stage; (2) the number and variety of core value subcomponents; (3) justifications for the assessment (the second and third assessments asked participants to justify their assessments for each species, with responses grouped into "information provided (only)", "personal or previous knowledge" or "both"); and (4) survey comments, transcribed discussions, and group dynamics (e.g., speaker dominance).

Results

A total of 84 individuals responded to the survey (27 US/CA scientists, 17 AU scientists, 27 US/CA managers, 13 AU managers). Summary of participant demographics are provided in Appendix B: Tables 3-6. As expected, many who completed the survey were unable to attend the workshop; results for survey questions are out of survey totals while those for the multi-stage consequence assessment are out of workshop totals. A total of 60 individuals participated in the workshop (21 US/CA scientists, 12 AU scientists, 13 US/CA managers, 14 AU managers), yielding a total of 33 scientists and 27 managers, and 34 US/CA and 27 AU participants. The results, when presented as percentages, are presented in the following order: US/CA scientists, AU scientists, US/CA managers and AU managers.

Information types

In the absence of peer-reviewed literature, participants most frequently rated 'supported/verified observations' as the first alternative source of information, followed by 'heuristic/expert observation/experience' and 'personal communication with scientist' (Table 3.3). Most participants chose 7-8 as the degree of change (based on a scale of 1-10, where 10 equates to higher quality) for a risk assessment that used non-scientific information, though the two Australian groups equally chose 7 and 3 (i.e., higher and lower quality, equally).

When provided with hypothetical combinations of uncertainty and strength of evidence for a high impact (both for the actual ANS under question and a species similar to the ANS under question), participants in all groups selected 'high' ratings more frequently in circumstances of strong observational/lay evidence than for uncertain experimental/scientific evidence. Other shared patterns included: similarity between the assessments of ANS and similar species (particularly for uncertain experimental/scientific evidence); and strong experimental evidence for the similar species received a 'high' rating more frequently than all combinations of evidence and uncertainty for the actual ANS (except with strong experimental/scientific evidence).

Table 3.3. Summary of responses to questions related to information type (out of 84).

When assigning impacts for nonindigenous species with absent or insufficient peer-reviewed impact data, it is appropriate to also include (choose all that apply):	**Frequency chosen**
Supported/verified observations (e.g. data from more than one person involved in resource management such as restoration planners or park director)	73
Heuristic/expert observation/experience	69
Personal communication with scientist	67
Lay knowledge (e.g. observational data from public such as port managers, long-term residents of a site, or fishers)	57
Impacts that are published but do not cite experimental analysis	47
Grey literature (e.g. websites, policy documents, databases, reports)	46
Incomplete and/or unfinished scientific studies	46
"Anecdotal" information, such as news stories	16
Unsupported/unverified observations	2

Attitude toward and knowledge of uncertainty (general)

Most participants in each group (100%; 88%; 93%; 92%) agreed with the existence of 'unknown unknowns' (i.e., uncertainty that is not or cannot be described). Slightly more participants in each group felt that few research questions could be answered with high certainty (56%; 59%; 48%; 38%) than felt that most research questions could be answered with high certainty (36%; 18%; 44%; 38%). However, many participants (92%; 82%; 96%; 77%) believed that with enough time and money, research can reduce these uncertainties and knowledge gaps. Most participants (92%; 88%; 78%; 84%) indicated that uncertainty is unavoidable but can be managed to provide reliable results. Other common sources of uncertainty identified by the participants in the open-ended questions included a lack of baseline knowledge, the effects of climate change, and predicting to what extent species will become invasive, particularly due to different ecological conditions (Table 3.4).

Table 3.4. Summary of: the biggest challenges to (C), and sources of uncertainty (U), in predicting future impacts of nonindigenous species, as identified by participants.

Research-based challenges and uncertainties	Total	Type
Human-mediated changing environment	**24**	**C**
Climate change	15	C
Overfishing, human structures and activities, inputs from land	1	C
Water quality and habitat change	2	C
Interaction with other ANS	1	C
Rate of introductions (as a challenge)	2	C
Uncertain behaviour/relationship with new environment	**17**	**U**
Adaptation	5	U
Time Lags	3	U
Long-term changes	2	U
Lack of baseline knowledge	12	U
Predicting what species will become invasive, how invasive	**26**	**U**
Due to physiological differences between genotype and within genus	3	U
Due to different ecological conditions (abiotic and biotic)	11	U
Unknown transport vectors	6	U
Spatial and temporal variability	7	U
Cryptogenesis	5	U
Uncertain inoculation conditions	2	U

Table 3.4 cont.

Research-based challenges and uncertainties cont.	Total	Type
Lack of understanding of impacts	**16**	**U**
Detecting impacts	4	U
Defining impact	2	U
Isolating impacts of a specific species from other factors	4	U
Differences in experimental methods	1	U
Policy-based challenges and uncertainties		
Risk assessment that is comprehensive yet accessible to policy makers	2	C
Controlling ANS without harming the environment	2	C
Type II errors (over-reaction to the threat)	2	U
Insufficient monitoring	2	C
Inaccurate modelling	1	U
Addressing uncertainty within appropriate timeframe	2	C
Credibility of risk and impact assessments	1	C
Research funding and support	7	C
Issue recognition	4	C

Uncertainty: Ambiguity

For scientists and AU managers, the ranked importance for protection of core values was (in descending order of importance) environmental, human health, economic and social/cultural (for US/CA managers, human health was ranked before environmental).

While environment and human health were the two most important values, there was an almost equal split between those who felt environment should be first, and those who felt human health should be first in the scientist and US/CA manager groups. The situation was similar for economic and social/cultural values (i.e., split in their ranking as the two least important core values). Australian manager participants, however, rated the environmental core value significantly more often than the other values (which were an almost-equal second).

Uncertainty: Knowledge Gaps

The results from the questions on knowledge gaps indicate that almost all participants (96%; 94%; 96%; 100%) felt that nonindigenous species have an impact due to their presence as a non-native component of the ecosystem, and of these, many (87.5%; 81%; 81.5%; 62%) felt that assigning a 'low' impact is appropriate if there is an absence of impact information for that species. The significance of knowledge gaps as a major cause of uncertainty was supported by the open-ended questions, as well; one of the most cited uncertainties by all groups to ANS impact assessment was lack of knowledge of impacts (Table 3.4).

Uncertainty: Systematic measurement error

Most participants (75%; 83%; 81%; 77%) agreed that avoiding Type II errors is more important than avoiding Type I errors when assessing ANS impacts. When data is associated with an insignificant *p*-value, many (67%; 59%; 81%; 77%) also felt this data may be used with discretion if no other data is available and about a quarter of participants (25%; 29%; 19%; 23%) felt it valid to use when assessing impact (only the scientist groups contained participants who felt the data was invalid, with 8% and 11% in US/CA and AU groups, respectively).

Uncertainty: Variability

When asked when past impacts are appropriate to use as predictors of future impacts for ANS, about half (43%; 65%; 50%; 55%) of participants chose 'most of the time'; and about half (54%; 35%; 50%; 45%) chose 'some of the time'; and only one group had participants that chose 'rarely' (US scientists, 4%). The participants cited several variability-related issues, namely, uncertain behaviour/relationship with new environment and spatial and temporal variability in the open-ended questions on uncertainty in and challenges to ANS risk assessment.

Attitude toward and knowledge of precaution

Most individuals felt that precaution is a necessary component of a risk assessment (91%; 88%; 85%; 84%); should be applied along a continuum, with greater potential threats requiring less certainty before taking precautionary measures (74%; 94%; 63%; 69%); and felt that the application of precaution included using all types of information (even non-scientific; 87%; 77%; 66%; 91%). Participants most frequently rated "in the final assessment, include even those species with low and/or unknown likelihood or low and/or unknown impact designation as possible risks" as a potential way to incorporate precaution into a risk assessment, followed by "when assessing impacts for a species using previously-documented impacts, use the impact of highest magnitude" (Table 3.5). Finally, most participants (with the exception of AU managers) felt that the provided WTO SPS Article 5.7 text on provisional action would suggest the use of precaution as a tool in risk assessment (92%; 71%; 92%; 45%).

Table 3.5. Summary of participants' views on steps to integrate precaution into a risk assessment.

Description of precautionary steps	Frequency chosen
In the final assessment, include even those species with low and/or unknown likelihood or low and/or unknown impact designation as possible risks	50
When assessing impacts for a species using previously-documented impacts, use the impact of highest magnitude	43
If impacts for a particular nonindigenous species are unknown, use impacts from a similar species with known impacts	42
Including public input regarding values and impact significance	32
Use conservative estimates when developing and/or using model parameters	32
Assume all cryptogenic species are nonindigenous; that is, if a species can't be determined to be native or not, assign non-native status	18
For nonindigenous species with unknown impacts, assign a "low" impact	18

When asked to rank core values according to the importance of applying precaution, both US/CA and AU scientists and US/CA managers ranked the core values differently than for overall importance, with human health most frequently ranked first, followed by environmental (AU managers retained the original rating, with environment ranked ahead of human health). As with overall importance, social/cultural and economic were ranked last in all groups, almost equally. This was supported by discussion amongst workshop participants. For example, an MBIC participant asked (and was answered in the affirmative by several participants), "Do you think that when it's a human health issue, as opposed to an environmental or social issue you tend to err toward the higher, rather than low? When there is high uncertainty."

Change in assessment with additional information and discussion

For all groups, the result of additional information and discussion was a net frequency of an increase in the assessed consequence (Figure 3.1; Appendix B5: Figures 1 and 2).

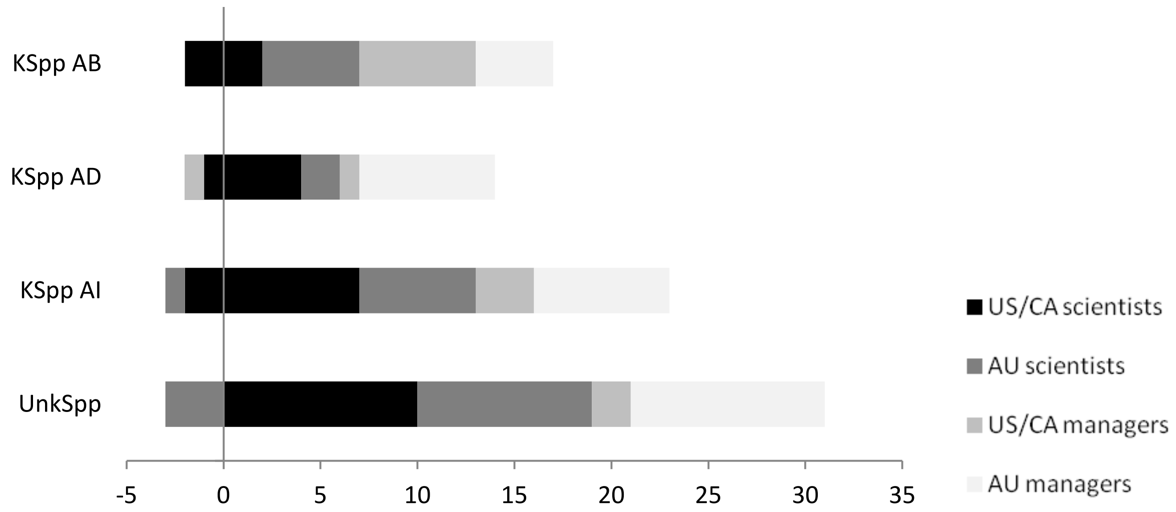

Figure 3.1. Number of species that saw a change in assessment for: known species after information (KSpp AI); known species after discussion (KSpp AD); known species after both (KSpp AB) and unknown species after discussion. Bars to left of zero indicate the number of species with lower consequence assessments; bars to right of zero indicate number of species with higher consequence assessments.

There was also an increase in the number of different subcomponents mentioned for each core value after information and discussion. Justification was used to examine the use of information. In addition to providing evidence for the participants' use of the provided information, there was slightly greater (proportional) use of personal/previous knowledge for social and cultural values (for all groups), and, to a lesser extent, the human health assessment. This may result from little or no impact information provided to participants for these core values, relatively (forcing them to rely on previous knowledge; Figure 3.2).

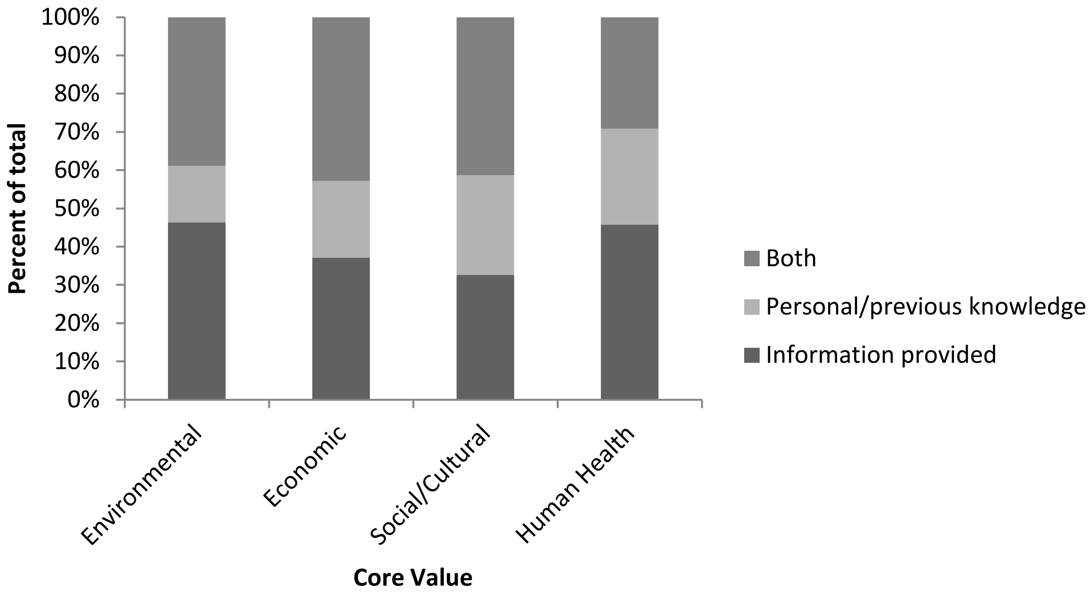

Figure 3.2. Summary of information used for each core value in the second consequence assessment.

Information use was then divided into whether the core value was a focus (i.e., whether there was impact information provided to the participants for that core value). The most noticeable trend was a general increase in personal or previous knowledge use when the species was not a focus, which may result from participants relying on their own guess or judgment (Figure 3.3). The reliance on any provided information for areas outside of the participant's expertise (i.e., human health) is also clear.

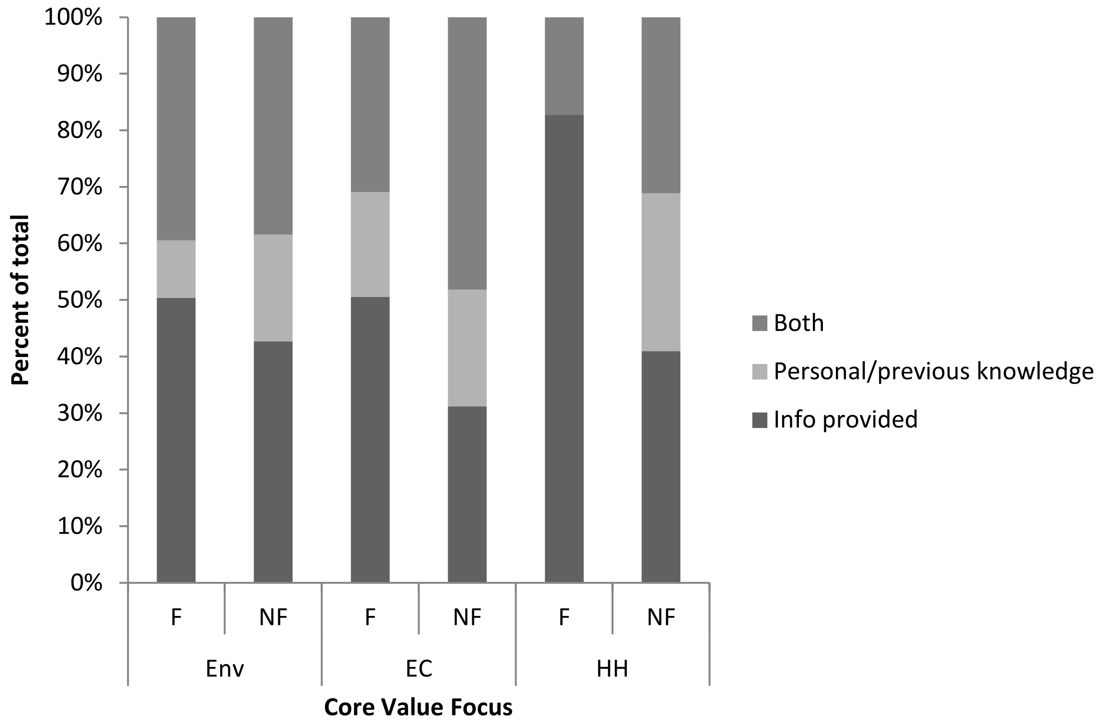

Figure 3.3. Use of information sources based on whether the core value was: the focus (F) of the information provided; or not the focus (NF). Env = environmental; Ec = economic; and HH = human health.

The modified Delphic process

Each workshop group broke into three smaller groups ('subgroups'), except for AU scientists (two subgroups). Of the eleven subgroups, review of conversation dynamics (based on number of comments, who was asked the questions and who led discussion) suggested three subgroups had two co-dominant leaders, three subgroups had one dominant leader and two subgroups had fairly even discussion. These leaders were generally male (one exception) with moderate (5-10 years) to high (10+ years) experience. The workshops were not designed to test the influence of gender or experience on group agreement or leadership, and as such, analysis of changes in assessment were not possible given the small number of woman-led groups, the unknown views of individual participants (the poster-based assessments were non-identifiable), and the difficulty in identifying participants in recordings. However, some influence of leaders in group dynamics was seen for two subgroups, in which the leaders had to leave part-way through the assessment. In the first case, a lesser-experienced (0-5 years) participant increased his contribution and in the second, a female participant assumed a moderate leadership role.

Analysis of discussion from each workshop revealed themes (Table 3.6) common to several of the groups. For all groups, the discussion that occurred during the Delphic process influenced the consequence assessment, as evidenced not only by the results from the consequence assessment (significant increases in consequence estimates), but also in its justification for choice of consequence level in the final assessment (e.g., "talks with scientists" and "expert info in group").

Table 3.6. Themes repeated in two or more workshops.

Major Theme	Description
Desire for unknown category	Several participants (particularly in US/CA scientist subgroups) were unwilling to assign impact without additional information, particularly scientific literature. For example, comments such as "would have liked another category, similar to 'not enough information to decide'", "I didn't want to be forced to make an assessment that I felt like I didn't have enough information for", and "I didn't want to have to peg, either 'low', 'medium', or 'high' – I wanted to have an 'unknown' [category]."
Associated economic and social/cultural values	Participants in all workshop groups perceived a relationship between economic and social/cultural (and, to a lesser extent, human health) consequence magnitudes (i.e., social/cultural effects were derived [directly or indirectly) from economic effects]. For example, ("I think of them [economic and social/cultural] as the same. If you reduce the fishery, it's a social impact"; "social and cultural and human health – flow-on from economic effects"; "social and cultural...tie slightly in with economic impacts" and "for social and cultural impacts I rated them about the same as [the species that] will affect boats"; and "economic, social and cultural kind of correspond to me, if it's high in economic, its high in the other – that's my reasoning, only because economic impacts kind of correlates with social impacts".
Location matters	Where the impact occurred influenced the perceived consequence. For example, "if it were in a pristine place or some place that I loved, I'd be 'extreme', 'pegged out', 'red-lined' every time". Also, in response to a poisonous species, an Australian joked that it "sounded like most of our native species".
Impacts are relative	Participants proposed using relative scales to assess consequences. For example, "every time you say 'high' it helps to think of all the other non-natives out there... let's think about green crab, zebra mussels..."; "I would like a scale – the 'worst' non-native and the 'least' impactful, and compare it to this scale" and "I think what's interesting is these are all relative scales and we haven't talked so much about how each of us has been deciding them relatively...I mean, when we were talking about lionfish, for me, even though they are obviously a hazard, in my mind, they are pretty low relative to other hazards."
Comfort zone and field-specific assessment	Several of the groups discussed how their field (and associated comfort with associated values) influenced their assessment. Specifically, several proposed that their expertise in the environmental field affected their assessment for the other values via comments such as, "Interesting – as marine biologists, we know a lot about environmental, but when it comes to social, economic, human health – if we had an economist, in this group, I'm sure it would be a lot different."

The implications of a comfort zone and field-specific assessment were discussed by a participant who pointed out that for species with little information, most in the group had unanimously "high" uncertainty, while some in the group had ranked "low" uncertainty only for social/cultural and human health core values. One participant justified this by explaining that economic effects may result from environmental impacts (i.e., potential economic impact hence high uncertainty), to which the first participant challenged "couldn't you say the same thing about social and cultural?" This conversation seemed to suggest that several of the participants assumed/assessed higher impact to economic core values, using methods that, when no direct economic or social/cultural impact was stated, assigned indirect effects to economic values while assuming that lack of direct information for social and cultural and human health values meant no impact. For example, "if it had a particular reproduction method, then it may have high economic or environmental impact, etc, but if it didn't state particularly a social impact...it had to be particularly stated." One participant offered an explanation for this difference: "these two are comfort zones – environmental and economic – what we've always dealt with. Social and cultural and human health, we've only just started to deal with."

Discussion

This chapter identified and mapped the presence of uncertainty, as well as attitudes toward uncertainty and precaution, within ANS risk and consequence assessment, with an expectation that participants from Australia and the United States, as well as scientists and managers, would have different views on these issues. In contrast to this expectation, there were few qualitatively significant differences between groups (differences are noted, where they exist). As such, results are summed and discussed across groups. The results of consequence assessment for ten ANS, completed by the survey participants over three stages, indicate that the provision of information and opportunity for group discussion via a Delphic process had a general effect of increasing the assessed consequence for each species. While the results provide several outcomes potentially useful for application in ANS risk assessment and management (discussed in "Recommendations", below), the limitations and potential biases also warrant consideration.

The sampling methodology (purposive, self-selecting) and size of this study presents some potential biases and limitations, respectively. While practical given the pre-identified target audiences and resource limitations, the use of non-probability sampling techniques can introduce biases into the study. For example, as this study included ANS experts attending conferences and required a fairly extensive time commitment, participants may have tended to be more ‘outgoing’ or ‘passionate’ about the issue, have more time (which may correlate with less experience) or be different in some way that makes them likely to go to conferences (e.g., work for federally-funded sources, which tended to have larger travel budgets during the workshop periods). Each of these may affect the respective consequence assessments (Webler et al. 1991). While many view probability sampling as less bias-inducing, it can have self-selection bias if participants of a certain demographic or other characteristic tend not to participate (Whitehead 1991, Braver and Bay 1992).

The small sample size restricted the potential power and therefore use of inferential statistical analyses. This latter issue could be addressed with greater sample size and regression techniques (based on interval measurement of experience; e.g., Donlan et al. 2010). This is difficult, however, due to the generally small numbers of experts in a given field. One potential improvement, given additional time and money, to decrease potential bias and increase sample size, would be stratified random sampling, in which additional venues are selected for sampling (e.g., agency workshops, or departmental or laboratory meetings), from which a random sample is chosen (Danz et al. 2005). However, given the small population size of ANS experts and similarity between the structure of the study and the structure of situations to which the study’s outcomes may be applied (i.e., workshops to determine consequence or risk, which commonly occur as part of such conferences), purposive sampling and the associated results and lessons learned from this study should be sufficiently valid for applications to biosecurity and ANS management.

A second potential bias results from the survey design, that is, the survey included questions with many agree/disagree-based Likert scale response lacking a “neutral” option and the consequence assessment provided participants with a Likert scale response lacking an “uncertain” or “unknown” option. Although the lack of a “neutral” option has been shown to effect response distributions (Bishop 1987), the benefits of removing a neutral option include a reduction of the effects of the social desirability bias (i.e., participants making a choice that will be viewed favourably by the surveyor or other participants; Garland 1991) and the directional bias Net Acquiescence Response Style (i.e., the tendency to show greater acquiescence, often via a movement of negative responses toward the neutral option; Baumgartner and Steenkamp 2001, Weijters et al. 2010). While not including an “uncertain” or “unknown” option (the desire for which was expressed by several of the participants and elsewhere; e.g., Morgan and Henrion 1990) has had mixed effects on response distributions (e.g., Li and Mattsson 1995, Alberini et al. 2003). Advantages include a reduction of “satisficing” (from “satisfy” and “suffice”), in which respondents choose not to expend energy making the optimal decision, instead merely making a choice that seems adequate (Krosnick 1999).

Characteristics of satisficing include less consideration of the question at hand, less comprehensive search of personal held knowledge, or less careful consideration of provided information; participant's may indicate "don't know" in order to avoid expending effort or making a risky judgment (Krosnick 1999). Given that real-world ANS risk assessment and management require some decision of consequence (as opposed to "unknown" or "uncertain"), albeit with an associated degree of uncertainty, I would suggest the absence of an "uncertain" category (with an allowance for participants to describe their uncertainty, separately) was useful in preventing satisficing and appropriate for the study. A similar approach that allows experts to indicate their uncertainty in order to overcome their reluctance to make a decision in the absence of sufficient information has been used by Marsh et al. (2007).

Despite these limitations, participants identified gaps in understanding of ANS as a significant challenge to ANS risk assessment. As a result of these knowledge gaps, participants expressed a desire for an "unknown" category to be included in assessment/answer options. However, from a management perspective, assigning a species "unknown" does not produce a useable outcome; in order to provide a policy-relevant risk assessment, experts need to provide some estimate of impact with an associated uncertainty rating. Outcomes of this study suggest that in these situations of high uncertainty, experts can still provide functional consequence assessments through the use of various information sources (empirical and non-empirical), a Delphic process that includes a variety of stakeholders, and precaution. In particular, group discussion and input benefits the risk assessment process and outcome, such as via increased diversity of opinion and highlighting common themes as well as false or extreme opinions (Krueger and Casey 1994). This effect and potential improvement via discussion is seen in other group exercises, when participants interact with and expand upon other participants' input to synergistically produce new forms and amounts of knowledge (Litosseliti 2003).

In addition to aiding decision-making in a nonindigenous species context, the modified Delphic process may aid this process in other areas of conservation. As the number of global threats to conservation (e.g., climate change and habitat modification) grows, so inevitably does the complexity of their interactions and the difficulty in understanding the synergies at play. As the rate of change will likely outstrip the rate of research, decision-making processes will be challenged with increase uncertainty. Conservation experts are already using Delphic processes to prioritize funding for protection of threatened species (Marsh et al. 2007). Knowledge of the performance of these practices adapted to mitigating uncertainty should be sought via empirical research and in situ practice in other fields, as well.

Recommendations

Based on the outcomes of this study I suggest the following recommendations for mitigating uncertainty within a consequence assessment:

- **When empirical evidence is lacking for a particular ANS at the desired spatial and temporal scale, alternative information sources can include empirical evidence from other regions or from similar species, as well as non-empirical evidence.** Participants advocated using a variety of information sources (even those that are 'non-scientific') to manage uncertainty and produce a consequence assessment and from this, a risk assessment (Table 3.3). Specifically, the study supported using strong observational or lay evidence over uncertain experimental or scientific evidence, personal communication with scientists; (the strongest) evidence from past impact studies; and evidence for species similar to the one(s) under question, with strong experimental evidence for a similar species trumping uncertain experimental or strong observational evidence for the species in question. Grosholz and Ruiz (1996) demonstrated similarity in ecological impacts of the European green crab, *Carcinus maenas,* across sites. Ricciardi and Rasmussen (1998) suggest using the assumption that invasive species will continue to invade new habitats as a basis for identifying species with an invasion history in order to predict new introductions. For example, the zebra mussel, *Dreissena polymorpha,* has had very similar impacts in both the US and Europe (where its invasion history dates back 200 years) (Ricciardi 2003).

  Evidence from physiologically or phylogenetically similar species may offer an alternative source in cases where no impact data exists at any scale (as is the case for many species) (Byers et al. 2002). Ricciardi (2003) provides the comparison of *Limnoperna fortunei* with *Dreissena polymorpha* as an example; while poorly studied in its native habitat, *Limnoperna* appears to be having impacts similar to *Dreissena* in their respective nonindigenous habitats. However, while supported by participants and several studies, these inter-species comparisons should be made with care. For example two similar gobiid fishes were introduced in a similar spatial and temporal context, with one (*Proterorhinus marmoratus*) remaining inconspicuous and the other (*Neogobius melanostomus*) invading new habitat and imposing impacts (Jude et al. 1995).

- **Using available evidence and exercising care to avoid potential biases, the modified Delphic process can improve the consequence assessment outcomes while identifying and mitigating the effects of uncertainty.** Despite a lack of extensive (or any) information, the group assessment, as well as the individual assessments and their justification, provided a 'best estimate' of consequence – an outcome that can be applied to real policy decision-making processes. Decision makers can also use the variability within the group assessment, as well as the participants' own assessed uncertainty, to judge the reliability of this estimate (Campbell 2008). The clear analysis and documentation of the rationale behind these decision-making processes (as provided by the study methods) will ensure the outcomes are transparent, as mandated by the SPS Agreement Article 5, as well as providing valuable insights into the values, norms and influences on expert judgment (e.g., the influence of location on perceived consequence and desire for 'relative impacts') (Dahlstrom et al. in review). These outcomes support other literature that have used individual or group (via the Delphic process) judgment, with considerations of uncertainty, to make decisions in the assessment process, potentially via Delphic process (e.g., Halpern et al. 2007, Marsh et al. 2007, Teck et al. 2010).

One potential bias associated with the Delphic process is the availability or recall bias. Recall bias occurs when participant response is affected proportionally more by recent events as opposed to full consideration of relevant facts. Group discussion may have influenced the final estimates. However, influence from other experts' input is an intended outcome of the Delphic process. Given that the participants had the impact literature available and were not influenced in a specified direction, this bias should not adversely affect the outcome. Related to the concern of directional influence is the influence of anonymity and group leaders. The lack of anonymity in the modified process allowed for a rapid cycle of assessment and discussion, which reduced the resources and participant time required, and did not likely effect the assessment. By itself, anonymity (or lack of) has not been shown to effect response quality, direction or "social desirability" in both sensitive and non-sensitive contexts (Fuller 1974, Stone et al. 1977, Wildman 1977, Singer 1978). However, group leaders have been shown to increase group consistency (Suter 1993).

The presence of mostly male leaders was unsurprising given the significant influence this factor has been shown to play in past studies, in the direction of traditional gender roles (e.g., Piliavin and Martin 1978). Expectation States Theory suggests that status beliefs (widespread cultural beliefs of the competence of one group over another, e.g., males over females) lead to men as the more dominant, influential and proactive group, and women as the more subordinate, supporting, and reactive group (Wagner and Berger 1997, Ridgeway 2001). Empirical research supports this theory. For example, in a study of classroom dominance in graduate students, Brooks (1982) found males to have greater frequency and duration of speech and more interruptive behaviour. In addition, other studies have found that males are more dominant than women in mixed-gender groups (Skillings et al. 1978). Similarly, both males and females tend to abide by male more than female opinions (Eisler et al. 1975) and men are more likely to be selected as leaders (Eagly and Karau 1991). Where females led or participated more in the workshop discussion, this may have been due to their greater experience and "practice" in task-based activities (Lockheed and Hall 1976).

The effect of experience has also been well documented. For example, otherwise-dominant males defer to more experienced males (Brooks 1982), potentially via "self-other performance expectations" (i.e., the value of what individuals perceive themselves as able to contribute, relative to others; Ridgeway 2001). The lower the performance expectations, which is often based on things such as titles and formal roles, along with the associated power and resources, the less likely individuals are to give input and the more likely individuals are to change their views to align with others.

- **To ensure adequate consideration and weight are given to all core values, the modified Delphic process should include a variety of stakeholder groups.** Several of the groups discussed a perceived connection between economic and social/cultural consequence levels and postulated that this may be due to the environmental-based expertise of the participants. That is, they may not have sufficient knowledge and understanding to properly judge the impact on other core values. This frequently self-acknowledged lack of expertise in non-environmental areas, as also evidenced by the discussion and background (only two participants had any non-science based education, one in public policy and the other in philosophy), is particularly important given the dependence on 'personal knowledge' displayed in the justification section. A study of fisheries scientists by Schwach et al. (2007) also identified this as a potential problem during the assessment process. The fisheries scientists felt they were being asked to provide assessments for issues beyond the scope of their knowledge base, analytical capabilities, or training (specifically, economic and social issues). Scientists were not willing to consider these "extra-boundary" issues in their analyses and called for increased identification and involvement of other relevant experts and stakeholders. While natural resource managers are described as more open to considering non-scientific information due to pressure to make practical decisions with imperfect information (Cullen 1990), they are not always willing to do so, either (Johannes 1989, Mackinson 2001).

Several of the participants in this study may also have reacted differently to the "extra-boundary" requests made of them. In several subgroups, familiarity with the environmental (and to a lesser extent, economic) core values may have led to the opinion that environmental values were the most important to protect and a difference in assessment when the participants were highly uncertain. For those areas where they were familiar, they could imagine a direct or indirect consequence from information, such as the species' reproduction method and from this assign low consequence with high uncertainty. In contrast, for those areas where they were unfamiliar, they required direct information on an impact in order to imagine a consequence, and as such, when direct information was not provided, assigned low consequence with low uncertainty. If this heuristic is indeed occurring, the treatment of human health may present a contradiction between attitude and behaviour; participants rated human health as the most important for applying precaution (and several participants affirmed this in the discussion), yet several participants seemed to apply less precaution in the actual assessment (i.e., low consequence with low uncertainty). Though not a universal trend among participants, this phenomenon certainly warrants additional attention – particularly given the influence of ambiguity in rating the relative importance of the core values (i.e., participants were split in which values they considered most important) (Westman 1985, Klinke and Renn 2002, Peel 2005), and is discussed further in Chapter 4.

The importance of including stakeholders with a variety of knowledge-bases is supported by Marsh et al. (2007), who included experts in community and indigenous values in their assessment of threatened Queensland frog species. In response to failings of the current North Sea cod management regime, Schwach et al. (2007) provide a fisheries-related example of public input via a Marine Stewardship Council, in which concerned citizens are provided a chance to interpret the science and provide their take on the relevant management strategies and consequences. In a biosecurity context, an organism impact assessment by Campbell (2008) captured and assessed information across core value expertise using various focus groups including social scientists, environmental managers, environmental consultants, economists, biologists, ecologists, indigenous peoples, and lay people. The success of these assessments in integrating opinions from a variety sources was largely achieved via the presence of an expert in risk assessment.

The stated importance of location also underscores the importance of using a variety of stakeholder groups, particularly if the assessment is region-specific. Several of the assessors indicated they would perceive a species to have a greater consequence if the impact was in an area important to them. This example of NIMBYism (the "not in my backyard" attitude that originated in response to environmental hazards; McGurty 1997) suggests that assessors far from the region under assessment may perceive the impacts to be of less consequence than those near the region. While it could be argued that close priximity could lead to a loss of objectivity, the perceived consequence must include those actually experiencing it. Thus, the best outcome may be achieved by a balance of local and distant stakeholders, whose assessments could be integrated by a trained risk assessor.

- **Precaution, which the participants identified as a necessary and valid tool in risk assessment, can be implemented in ways that support conclusions and recommendations from other components of the exercise.** In addition to participants' stated views on the necessity of integrating precaution into a risk assessment, their views on the importance of avoiding Type II errors (over Type I errors) and of considering even 'non-significant' results (which underscores the importance of power analysis for ANS studies) also supports the initial endorsement of precaution. The two most frequently identified methods to incorporate precaution into a risk assessment were 'include even those species with low and/or unknown likelihood or low and/or unknown impact designation as possible risks' and 'when assessing impacts for a species using previously-documented impacts, use the impact of highest magnitude'; when information exists, use the worst case scenario and when no information exists, assign at least a 'low' impact (with an associated high level of uncertainty) to keep the species in the assessment and available for updates with new evidence. Including species with low and/or unknown likelihood or impact does not become overly cumbersome to the risk assessment and risk management process due to the realities of biosecurity efforts. That is, biosecurity agencies often have limited budgets and will thus generally focus management or restriction efforts on those representing moderate to extreme risk. Including the species with unknown and/or low likelihood or impact thus does not necessitate actively managing these species, but ensures these species are not forgotten by management, as well as facilitating updates to the assessment as new knowledge is obtained and entered into the existing calculations. Additional discussion and considerations for determining the acceptable levels of impact (via effect size and Type II error rates and costs) are provided in Chapters 5 and 6.

The endorsement of precaution and its implementation via means described above, as a response to uncertainty by expert scientists and managers, suggests these elements do not stand opposite or in deliberate ignorance of the scientific process, but are consistent with and direct responses to lessons from invasion biology. For example, some of the most well-known and injurious nonindigenous species (e.g. the zebra mussel, *Dreissena polymorpha* or the northern Pacific seastar, *Asterias amurensis*) were not known or predicted with any degree of (un)certainty. A participant with significant ANS research experience identified the extreme difficulty of predicting invasive species' impacts as potential justification for using precaution, i.e., the potential implications of "not knowing what we don't know [are too great]...we have repeatedly had examples of not being able to predict bad consequences, but we still believe that we can (predict consequences)."

- **Given this science-based endorsement by both scientists and managers, it may be appropriate to change the way decisions based on these tools are described.** For example, instead of labelling those tools described above as "precautionary", clarify that these tools are based on solid scientific opinion and practice. This latter emphasis has implications for developing biosecurity measures in line with the SPS Agreement; if the "precautionary" approaches (above, and in other literature) are identified not as "precautionary-based", but as scientifically valid and supported components of a risk assessment, this could facilitate non-controversial use of precaution in an SPS measure. This is supported by Bohanes (2002), who states that although the precautionary principle is too vague to be a stand-alone objective and transparent policy tool, its wider application in the WTO SPS Agreement could occur through procedural (as opposed to substantive) requirements for SPS measures. Peel (2005) also supports the use of precaution through process, particularly in the presence of uncertainty. This development is also supported by participants' views that the text of SPS Agreement Article 5.7 suggests the use of precaution in risk assessment.

Conclusion

These results (while preliminary) indicate that the group discussion and other input produced by the Delphic process improved the risk assessment process and outcome. The strength of the recommendations outlined above increases when considering the effects of combining the Delphic process with the results of the survey. For example, an alignment of attitude and behaviour as inferred by the increase in consequence ratings that suggests participants initially assumed no (or lower) consequence, which goes against their support for precaution in a risk assessment context. This outcome supports previous studies that found the Delphic process appropriate for discussion to clarify understanding and reduce uncertainties in consequence assessment (Webler et al. 1991). Other outcomes of the study lack the detail and dimension necessary to attribute causes and consequences, but warrant additional study. Australian managers, in particular, showed slightly different attitudes to the relative importance of core values and the role of precaution in the WTO SPS Agreement.

While this study was limited to environmental expert groups, use of the Delphic process with other groups will allow a multidisciplinary yet relatively standardised approach to consequence assessment, two important characteristics of risk management (Ward 1978, Klinke and Renn 2002, Petrosillo et al. 2009). Despite the promising outcomes of such an approach, particularly in mitigating the widespread uncertainty, challenges to effective risk assessments as a decision-making tool for aquatic biosecurity policy remain; the clarion call for more knowledge of ANS impacts remains as strong as ever.

CHAPTER 4. THE ROLE OF UNCERTAINTY AND SUBJECTIVE INFLUENCES ON CONSEQUENCE ASSESSMENT BY AQUATIC BIOSECURITY EXPERTS

Manuscript in review:

Dahlstrom, A., C. L. Hewitt, and M. L. Campbell. 2011. The role of uncertainty and subjective influences on consequence assessment by aquatic biosecurity experts. Journal of Environmental Management (in review).

Dahlstrom, A (60%), Hewitt, CL (20%), Campbell, ML (20%)

- CL Hewitt and ML Campbell both contributed to the idea, its formalization and development, and assisted with refinement and presentation.

Introduction

Uncertainty pervades the understanding of many environmental issues (Peel 2005), including the threat posed by ANS to environmental, economic, social, cultural and human health values (Parker et al. 1999). This uncertainty can result from gaps in knowledge, measurement errors (flawed measurements or models) and variability (effects of time and space) (Klinke and Renn 2002). Given these diverse forms of uncertainty, risk assessments often use input from expert judgment to determine consequence (Halpern et al. 2007) and risk estimates (Campbell and Gallagher 2007, Campbell 2008).

In addressing uncertainty, scientists and other experts are considered rational, objective assessors of risk (DuPont 1980, Fischhoff et al. 1984). However, this view has changed due in part to the growing appreciation of the subjective nature of decision making under risk and ignorance[8] (hereafter, DMURI) as well as using norms, values and other subjective inputs (Tversky and Kahneman 1974, Slovic et al. 1977, Pidgeon et al. 1992, Slovic 1999). This subjective view of risk is now often accepted in risk literature and by many risk-management agencies (e.g., Ferguson et al. 1998, Chisholm 2005, Standards Australia 2009). With it has come recognition of the importance of understanding the values, attitudes, norms and biases (herein, subjective influencing factors or SIFs) that impact information processing and decision making within individuals and agencies responsible for estimating and managing risks (Fairbrother and Bennett 1999).

Given the inherent and often substantial level of uncertainty associated with data underpinning risk assessment, the influences of SIFs potentially play a major, yet largely unrecognized, role in DMURI within a risk assessment context (Slovic 1999, Byrd and Cothern 2000). The role of SIFs in DMURI tends to be unrecognized or ignored for several reasons. SIFs are not easily identified or understood and are therefore rarely incorporated into risk assessments, making many experts unwilling or hesitant to formally acknowledge them. This inherently denigrates the value of SIFs due to their very subjectivity (Shrader-Frechette 1985, Plough and Sheldon 1987). The widespread influence of these SIFs, however, means they generally affect how individuals make decisions (Slovic et al. 1977, Fischhoff et al. 1984). Ignoring SIFs may cause conflict when combining various experts' judgments to provide a final risk estimate. As each stage of an assessment includes subjective judgments – from the identification of hazards and the selection of data to assess them, to the choice of assessment

[8] In this DMURI context "risk" and "ignorance" are defined in the manner of decision theory, i.e., known and unknown probabilities, respectively. Both indicate the presence of uncertainty. "Risk", when used separately from decision making, is defined in a biosecurity context.

methodology – understanding the relevant SIFs comprises an important step in an effective risk assessment (Einhorn and Hogarth 1981).

Despite the potential "errors" resulting from expert use of cognitive heuristics[9] and biases, expert judgment is considered a valid method for reducing uncertainty through deliberation and evaluation of evidence (Einhorn and Hogarth 1981) and is a necessary component of risk assessment due to the frequent absence of direct data (e.g., Halpern et al. 2007, Teck et al. 2010). One method to increase reliability of expert opinion is via group discussion and assessment, such as the Delphic process (Burgman 2005), or focus groups (Krueger and Casey 1994; Morgan 1997; Rabiee 2004). As discussed in Chapter 3, the Delphic process aids in choosing consequence values, obtaining information and filling data gaps by gathering opinion and beliefs about a hazard (often supported by evidence to ensure transparency) from a variety of groups, such as scientific and technical experts (researchers), representatives of relevant government agencies (e.g., fishery managers, government researchers, environmental agencies), stakeholder/community groups, industry, recreational and conservation organizations, and/or indigenous groups.

Understanding DMURI: rationally maximize utility or take the shortcut?

Decision Theory is an interdisciplinary subject that includes a variety of theories, all with the intent (direct or indirect) of explaining and predicting how individuals should and do make decisions under risk or ignorance. Decision making under risk (known probabilities) focuses on maximizing expected value and utility (where utility indicates the value of an outcome from the view of the decision maker; Peterson 2009). Decision making under ignorance (unknown probabilities; similar to Knightian uncertainty, in which probabilities are unknowable; Knight 1921) often uses the Maximin Principle, which posits that one should maximize the minimal value possible for each alternative (i.e., choose the alternative that produces the best of the 'worst' outcomes) (Peterson 2009). Expert judgment of ANS consequence is often analogous to decision making under ignorance, with unknown probabilities and consequences. Later research on decision-making looked to more psychological explanations for behaviours and the decision-making process (Slovic et al. 1977). For example, the Theory of Reasoned Action (TRA) and its modified successor, the Theory of Planned Behaviour (TPB), posit that three constructs (attitudes, subjective norms and perceived behavioural

[9] Heuristics are learned, declarative or procedural knowledge structures stored in memory (e.g., "rules of thumb", judgmental shortcuts, biases, educated guesses, intuitive judgments, or simply common sense tools) that have been learned and internalized by the individual (e.g., 'length implies strength') to deal with an increasingly complex world, in which individuals are forced to make decisions using either an overwhelming or insufficient amount of information (Chaiken et al. 1989, Chen and Chaiken 1999).

control surrounding an action) combine to produce behavioural intention (i.e., decision to perform an action) (Ajzen and Fishbein 1977).

Empirical research, however, proved these normative theories often did not accurately predict decisions and behaviours (Kahneman and Tversky 1979). Attention shifted to descriptive models such as the Heuristic-Systematic Model (HSM) to explain how individuals make decisions under risk and ignorance (Trumbo 1999). The HSM identifies two methods by which people make judgments: systematic processing (a comprehensive analysis) and heuristic processing (a shortcut-based analysis; this occurs if an individual is unwilling or unable to take the time or make the effort to carefully consider the evidence) (Chen and Chaiken 1999, Trumbo 2002). A variety of factors can influence heuristic processing, including personal SIFs as well as external attributes of the event. For example, the availability heuristic is a common heuristic by which individuals perceive an event as being more probable if it is easily recalled due to a recent or high-profile occurrence (Slovic et al. 1979). The recognition heuristic is a “fast and frugal” heuristic by which individuals rank an object as more fitting of a criterion based on their recognition of it (e.g., ranking a city as greater in population based on recognizing its name) (Goldstein and Gigerenzer 2002). Descriptions of heuristic accuracy are mixed. Some research has found that not only do heuristics allow an individual to make some sort of decision under uncertainty (Fischhoff et al. 1982), but also that the subsequent decisions accurately predict the outcomes, particularly when characteristics of cognitive strategies align with those of the associated environment (Goldstein and Gigerenzer 2002, Hogarth and Karelaia 2007). Yet other studies have found heuristics may lead to incorrect perception or judgment of the information (Tversky and Kahneman 1974, Slovic et al. 1979, Oppenheimer 2003).

Typically, if insufficient time, knowledge or motivation leads to a heuristic processing approach, the choice of a heuristic occurs based on several factors (Chaiken 1980, Chaiken et al. 1989, Trumbo 1999). The applicability of the heuristic is a factor based on the degree of relatedness and relevance between the decision and the heuristic, specifically whether one would consciously associate the heuristic to the decision in other circumstances (Chaiken 1980). The history of the heuristic (how often and how well the heuristic has been used in the past) is another factor that dictates use. The more frequent the heuristic’s use, the more likely the individual will use it again (Chen and Chaiken 1999). In addition, the ease with which a heuristic is recalled and applied may, to the individual, increase the perceived applicability of the heuristic and associated confidence in applying it to the judgment in question (Tversky and Kahneman 1974). This mental association tool used to make a decision for a specific judgment or situation has been termed the “specific-practice effect” (Smith 1990). As linkages may be predictable according to the judgment or situation and the group to which the judger belongs, understanding these associations will allow increased understanding and (if necessary) moderating or eliminating the influence of these heuristics (Chen and Chaiken 1999).

A final tool for predicting decision-making under (and particularly, perception of) risk and ignorance is based on demographic characteristics, such as exploring the role of characteristics such as gender, ethnicity, education, and nationality. For example, place of residence has been demonstrated to play a role in how risk is perceived (Masuda and Garvin 2006). In a study of the risk of red tides, Kuhar et al. (2009) found that Florida residents (as opposed to visitors) saw the red tides as being longer in duration, more frequent in occurrence, and more severe in impacts. In a study of threat to endangered sea turtle species, Donlan et al. (2010) found experts who worked on a particular threat type ranked it higher than those who worked on other threats. Other studies have shown the opposite. Peterlin et al. (2005) showed that Port of Koper (Slovenia) employees ranked threats of air, marine and noise pollution lower than the general public. Kivimaki and Kalimo (1993) showed nuclear power plant employees felt the safety of and likelihood of an accident in nuclear power plants to be less than the general public, with the latter risk negatively correlated to “commitment to the organization”.

Assuming guilt or innocence: a potential heuristic for ANS consequence assessment

Given the high uncertainty associated with ANS due to poor data quality and availability, expert decisions of ANS consequences may include the use of heuristics. While there are various potential heuristics that may effect this decision, experts in an aquatic biosecurity consequence assessment context are faced with a choice between two contrasting approaches (in a sense, heuristics) when information is lacking or scarce. Experts can assume “guilty until proven innocent” (herein, the “precaution” approach; Figure 4.1a) or assume “innocent until proven guilty” (herein, the “hindsight” approach; Figure 4.1b). These two options are subject to much debate in many disciplines. A brief background and presentation of these arguments for and against each approach are presented below.

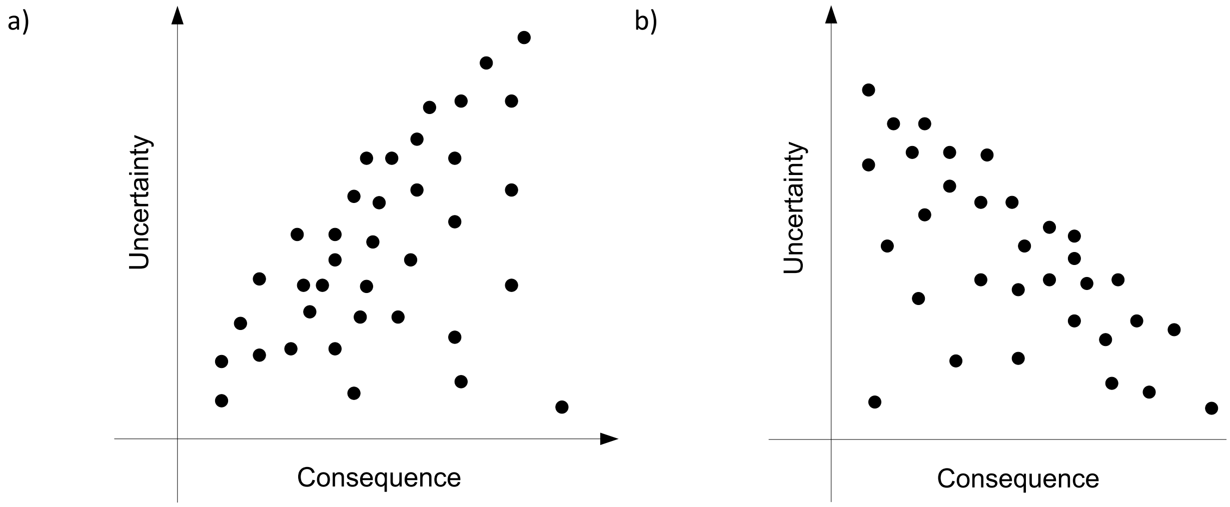

Figure 4.1. The hypothetical approach to assigning consequence in the presence of uncertainty under a: a) precaution approach, assuming "guilty until proven innocent"; or b) hindsight approach, assuming "innocent until proven guilty".

Precaution tends to make a "guilty until proven innocent" assumption when triggered by uncertainty (e.g., lack of available information). While espoused in many conservation-based agreements (e.g., CBD), many of the most powerful trade-related agreements do not acknowledge the application of precaution for environmental reasons, instead applying it to the protection of free trade (Chapter 2; Campbell 2001, Campbell et al. 2009).

While lacking a widely-recognized, pithy moniker, the approach that espouses "assigning no impact unless available data indicates otherwise" is the traditional and assumed approach to empirical scientific research and, within this, impact studies. This approach (herein termed the "hindsight" approach) potentially originates in null hypothesis significance testing. This statistical method is almost universally used by scientists (particularly in the fields of biology and ecology; Weinberg et al. 1986) to determine if results should be accepted as different from the null hypothesis or not. Basically, this approach assumes no difference between the treatment and control groups unless the statistical test yields a probability value that is below a pre-determined threshold (Lehmann and Romano 2005). In an impact assessment context, this translates into assuming no impact unless evidence data analysis proves otherwise or an "innocent until proven guilty" approach. The WTO adopts this approach. For example, the WTO SPS Agreement allows imports of all organisms unless specifically regulated based on sufficient scientific information (Takahashi 2006). Due to this prioritization of free trade over protection from nonindigenous species, and the overall paucity of ANS impact data, many claim the SPS Agreement is insufficient to keep out introductions (Jenkins 1996, Bright 1999, Campbell 2001, Riley 2005). At the individual level, the hindsight approach is often the assumed *status quo* for expert impact assessment, suggesting it may be the default heuristic in the absence of sufficient ANS impact data (Schrecker 1995).

Recognizing subjective influences on DMURI to ensure transparent risk assessment and policy

Both scientists (Burgman 2005) and managers (Williams et al. 2008) are subject to the influences of uncertainty and SIFs, factors which are rarely evaluated. This may have serious implications for risk assessment given the role these experts' risk estimates play in making management decisions. The decisions that result from these estimates are rarely evaluated in terms of uncertainty and SIFs. Empirical work has focused on estimating uncertainty surrounding consequence estimates as opposed to focusing on the influence of uncertainty and associated SIF's on those consequence estimates (e.g., Geneletti et al. 2003). Policy application also emphasizes the importance of identifying and tracking the "*post hoc*" uncertainty throughout the assessment process, as represented in the 1996 US Aquatic Nuisance Species Task Force Risk Framework (Lein 1989).

Some scientists are calling for a more systematic consideration of uncertainty. For example, in a study of fisheries scientists, Schwach et al. (2007) found a call from scientists to move uncertainty from the fringes to the heart of the scientific process. Due to the increased acceptance of expert risk attitudes and assessment as including subjective judgment, particularly when individuals are working at the limits of their expertise or have limited information or time (Slovic et al. 1979, Fischhoff et al. 1982, Fischhoff et al. 1984, Slovic 1999), it is feasible to assume that experts' basic cognitive functions are prone to biases (Tversky and Kahneman 1974, Freudenburg 1992, Slimak and Dietz 2006). In addition, these judgments are becoming increasingly common and unavoidable given the requests from policy makers, when developing biosecurity policy, of scientists for judgments and recommendations regarding the risk, regardless of the available knowledge. An increased understanding of how experts make these decisions under risk and ignorance will allow a more transparent risk assessment process and acceptable outcomes, particularly in situations of data scarcity that demand significant expert and stakeholder input. Thus, it is important to understand if and how these cognitive heuristics affect expert DMURI, and specifically, consequence assessment.

I explored the presence and effects of uncertainty on consequence assessment with four groups of ANS experts, to further understand the use of several heuristics and biases. Specifically, I investigated the use of the precaution versus hindsight approach, and the effect of species distribution, species name and core value on the expert consequence assessment for 10 ANS (the same 10 ANS introduced in Chapter 3; 5 real and 5 theoretical species). As stated in Chapter 3, I chose to conduct the exercise in the US and Australia because both countries have extensive biosecurity programs that would ensure sufficient workshop participation. These two countries also have a suite of differences including: their implementations of precaution; experience of invasion histories; and national attitudes and norms to quarantine and biosecurity. These similarities and differences will facilitate comparisons. For example, the US government has rejected international instruments that integrate the precautionary principle (e.g., the Convention on Biodiversity Protocol and the associated Biosafety Protocol) (Shaw and Schwartz 2005) and taken a more WTO-based stance toward precaution (e.g., stating that any decisions using precaution must be based on costs and benefits, as well as significant scientific evidence) (Cameron 2006).

Conversely, the Australian Environment Protection and Biodiversity Conservation (EPBC) Act (1999) affirmed Ecologically Sustainable Development and with it, the precautionary principle (Stein 2000). This contrast suggests differences between the target groups in the application of precaution (as well as differences by core value), with Australian participants expected to tend toward a precaution approach (Figure 1a) and US participants expected to tend toward a hindsight approach (Figure 1b) when making decisions under uncertainty. The differentiation between scientists and managers will also allow comparison of the respective application of precaution in decision-making under uncertainty between expert types. As scientists tend to follow a frequentist approach focused on avoiding falsely assigning an impact, I hypothesized that their response will reflect the hindsight approach. As managers are more often held responsible for the often high cost (ecological and economic) of missing a threat to natural resources, I hypothesized that their response will reflect the precaution approach. The different invasion histories allow comparisons of the effect of distribution on consequence magnitude (e.g., Australia contains species the US does not, and vice versa, which may affect their respective ratings).

Methods

Data was collected using survey and group discussions, using methods described in Chapter 3. Chapter 3 focused heavily on the inputs to and process of risk assessment. It used participant's views on components such as information sources, precaution and uncertainty, as well examining the effectiveness of a modified Delphic process, with a view of improving the process. Two of the strongest themes identified in this analysis were the participant's strong recognition of the presence of uncertainty and support for the use of precaution. Also evident was an increase in consequence magnitude post-information and discussion. The methods in this chapter aim to explore these trends further, focusing on the relationship between uncertainty and consequence. Understanding the more subjective decision-making processes involved in assessment may also help explain the change in assessment magnitude seen in Chapter 3.

Statistical analyses

Several descriptive statistical methods were used. Scatter plots were used to show consequence versus uncertainty ratings from each workshop. Radar graphs were used to show mean consequence values of each core value (dependent variable, represented by multi-coloured lines on the chart) for each species (spoke ends of radar graph) both for stages two and three of each workshop (one radar graph for each) and the mean consequence values of each workshop (dependent variable, represented by multi-coloured lines on the chart) for each species (spoke ends of radar graph) for stages two and three of each core value (one radar graph for each).

Given the ordinal nature of the data, it was necessary to perform inferential statistical analyses using nonparametric methods. Spearman's rank correlation, Mann-Whitney U-Test and Kruskal-Wallis tests were used to analyse the data. Spearman's rank correlation coefficient was used to explore any correlation between consequence and uncertainty. The Mann-Whitney test compared the consequence ratings for *C. scalpelliformis* and Unknown Algae, as well as for *B. ostreae* and Unknown Parasite, to test for an effect of species name and core value, respectively (given these two species pairs were the only pairs with quantitatively equivalent described impacts). For these analyses, the original consequence categories (negligible, low, moderate, high, extreme; 1-5) were combined into three categories (low, moderate, high; 1-3) to increase power. The Kruskal-Wallis test compared consequence ratings for each core value of each species, with the workshop groups as the grouping variable, to determine differences between workshops. In addition, the Kruskal-Wallis test compared consequence ratings for the 'focus' core value for each species (those without a 'focus' value were assessed for all core values) with the country of origin as the grouping variable, to determine if species' presence in assessing country affected assessment. The Friedman test was used to assess differences between all four groups in the mean consequence ratings for each species to determine how the groups varied, overall. The Wilcoxon Signed-Rank Test was used for post hoc analyses. Given arguments for the overly conservative outcomes of adjustments for multiple comparisons (e.g., the Bonferroni adjustment), no adjustment was made when determining differences between individual workshop groups (Rothman 1990). The use of nonparametric methods precluded the determination of power for non-significant results (Faul et al. 2011). However, the potential values and implications of associated power are discussed in the Discussion.

Results

There was a statistically significant difference in overall (mean) consequence ratings between workshop groups for the second assessment, for all core values (environment: $\chi^2_{[10]}$=11.16, p=0.01; economic: $\chi^2_{[10]}$=19.091, p>0.001; social/cultural: $\chi^2_{[10]}$=14.724, p=0.002; and human health: $\chi^2_{[10]}$=13.320, p=0.004), with the primary differences between AU managers and other groups (Table 4.1). No statistically significant difference occurred between workshop groups in the third assessment. A summary of impact values by core value and workshop group are shown for each species in Figures 4.2-4.4.

Table 4.1. Differences in overall mean consequence ratings, by workshop group. Sc=scientist; Mg=manager; AU = Australia; USCA = United States/Canada.

Core Value	Comparison	Z	*P*
Environmental	AU Sc/USCA Sc	-2.191	0.028
	USCA Sc/USCA Mg	-1.376	0.169
	AU Mg/USCA Sc	-2.805	0.005
	AU Sc/ USCA Mg	-0.153-	0.878
	AU Sc/AU Mg	1.070	0.285
	AU Mg/USCA Mg	-1.682	0.093
Economic	AU Sc/USCA Sc	-1.581	0.114
	USCA Mg/USCA Sc	-1.478	0.139
	AU Mg/USCA Sc	-2.803	0.005
	AU Sc/ USCA Mg	-1.125	0.260
	AU Sc/AU Mg	-2.346	0.019
	AU Mg/USCA Mg	-2.803	0.005
Social/cultural	AU Sc/USCA Sc	-1.784	0.074
	USCA Mg/USCA Sc	-0.059	0.953
	AU Mg/USCA Sc	-2.599	0.009
	AU Sc/ USCA Mg	-1.599	0.110
	AU Sc/AU Mg	-0.255	0.799
	AU Mg/USCA Mg	-2.599	0.009
Human health	AU Sc/USCA Sc	-0.561	0.575
	USCA Mg/USCA Sc	-1.886	0.059
	AU Mg/USCA Sc	-2.191	0.028
	AU Sc/ USCA Mg	-1.683	0.092
	AU Sc/AU Mg	-2.293	0.022
	AU Mg/USCA Mg	-2.803	0.005

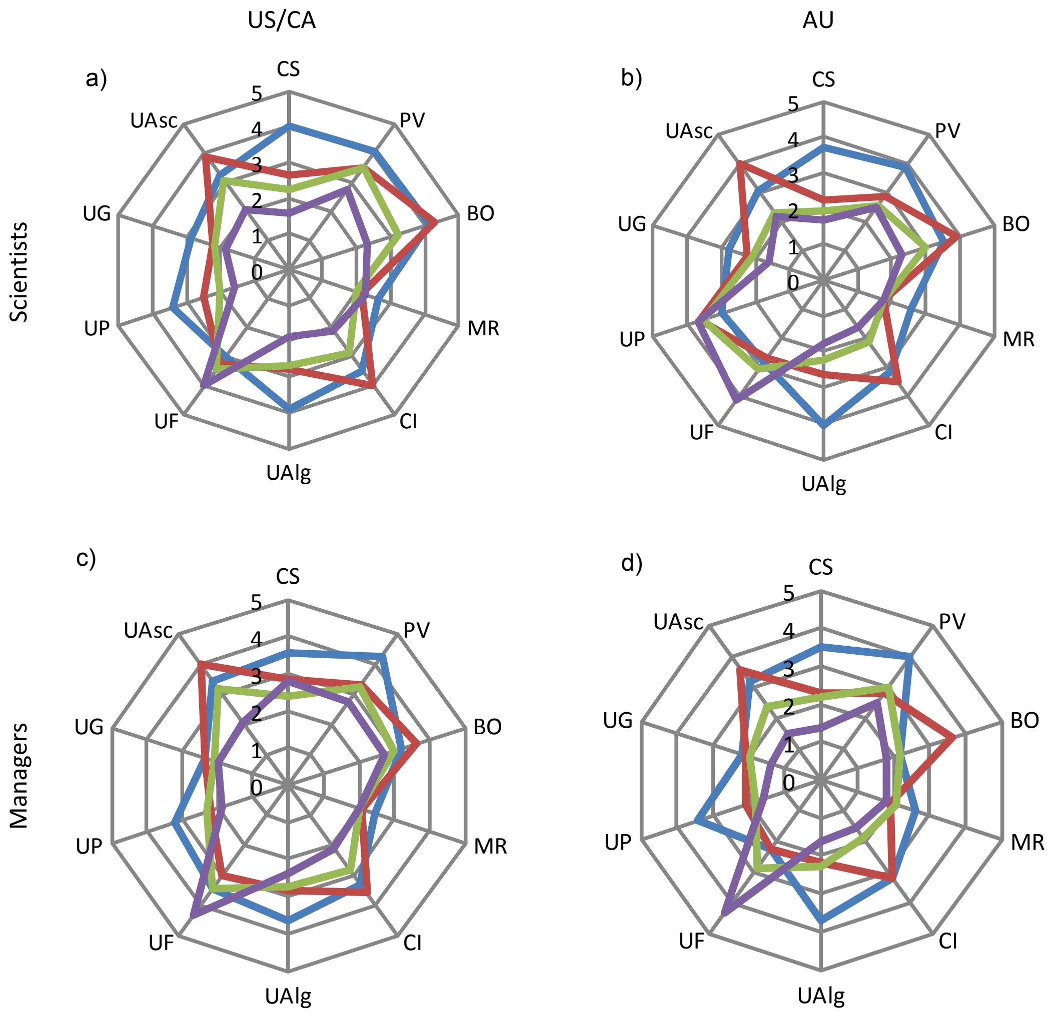

Figure 4.2. Mean consequence assessments for each core value at stage two of each workshop, with: a) US/CA scientists, stage two assessment; b) AU scientists, stage two assessment; c) US/CA managers, stage two assessment; d) AU managers, stage two assessment. Blue=Environmental; Red=Economic; Green=Social/cultural; and Purple=Human health. Each species is a spoke of the graph, with corresponding species directly opposite each other via the line through centre: CS=*Caulerpa scalpelliformis*, UAlg='Unknown Algae', PV=*Pterois volitans*, UF='Unknown Fish', BO=*Bonamia ostreae*, UP='Unknown Parasite', MR=*Maoricolpus roseus*, UG='Unknown Gastropod', CI=*Ciona intestinalis*, and UAsc='Unknown Ascidian'.

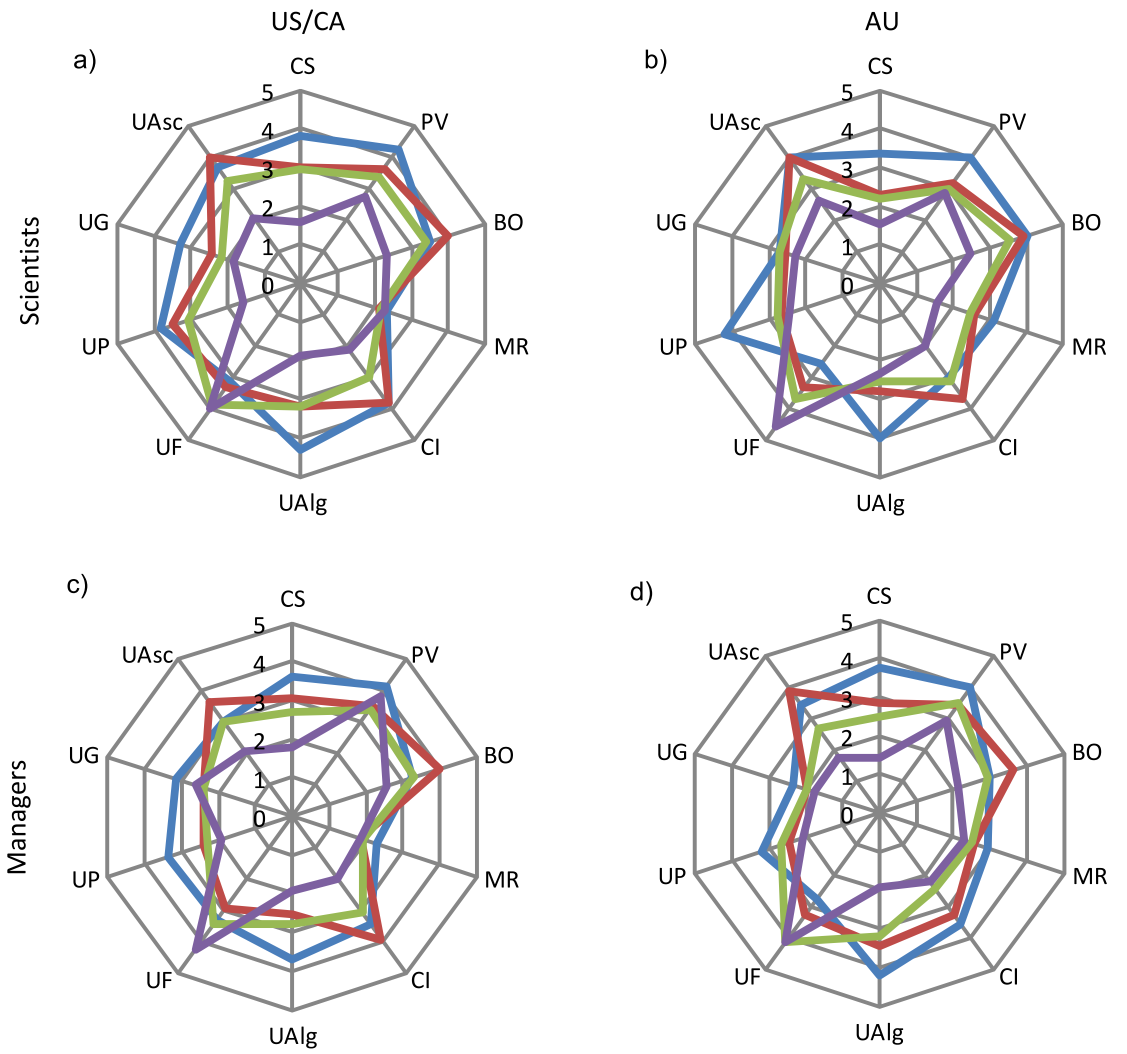

Figure 4.3. Mean consequence assessments for each core value at stage three of each workshop, with: a) US/CA scientists, stage three assessment; b) AU scientists, stage three assessment; c) US/CA managers, stage three assessment; d) AU managers, stage three assessment. Blue=Environmental; Red=Economic; Green=Social/cultural; and Purple=Human health. Each species is a spoke of the graph, with corresponding species directly opposite each other via the line through centre: CS=*Caulerpa scalpelliformis*, UAlg='Unknown Algae', PV=*Pterois volitans*, UF='Unknown Fish', BO=*Bonamia ostreae*, UP='Unknown Parasite', MR=*Maoricolpus roseus*, UG='Unknown Gastropod', CI=*Ciona intestinalis*, and UAsc='Unknown Ascidian'.

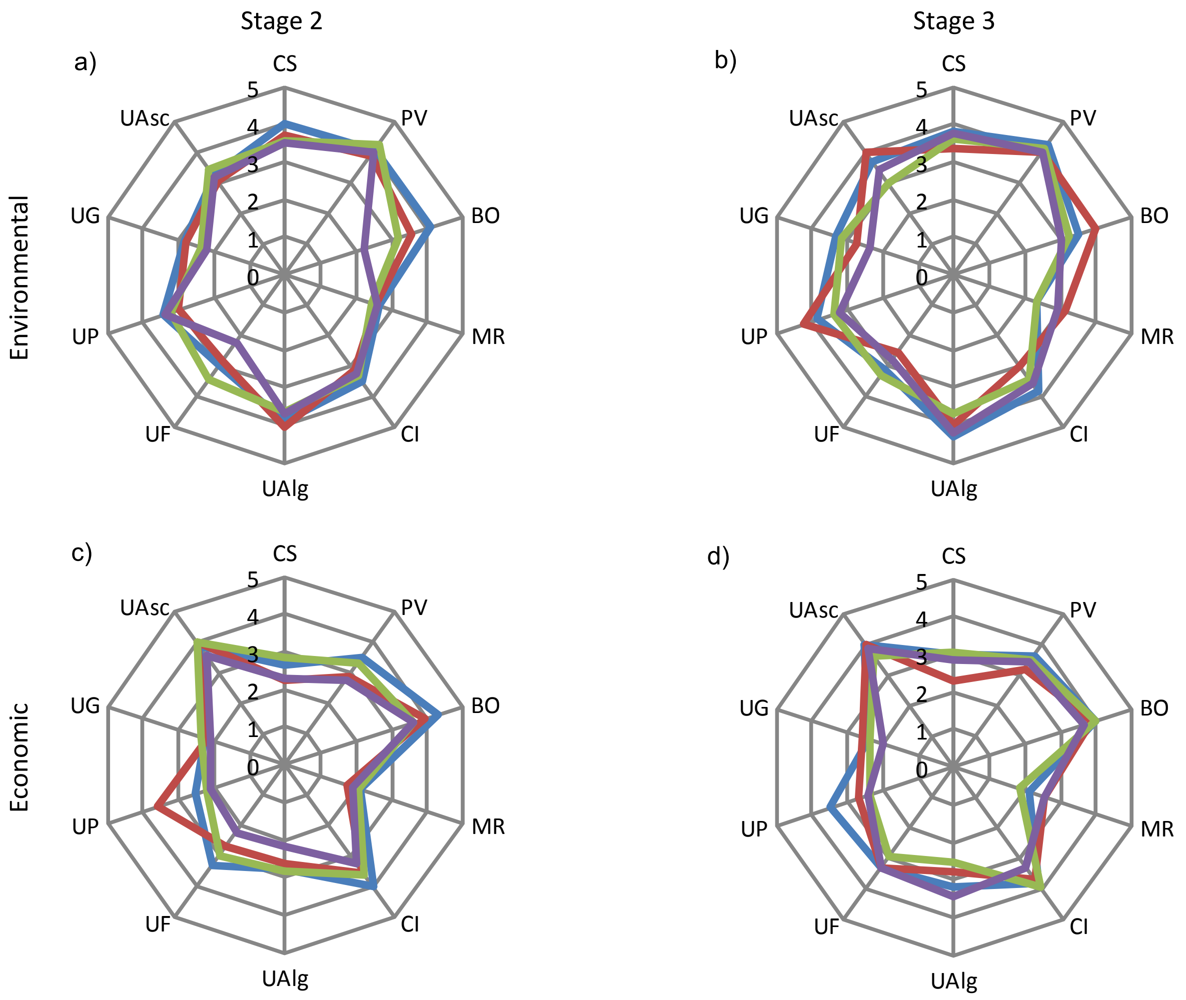

Figure 4.4. Mean consequence assessments of each species for each workshop group at stages two (2) and three (3) and for each core value, with: a) Environmental, stage 2; b) Environmental, stage 3; c) Economic, stage 2; d) Economic, stage 3; (overleaf) e) Social/cultural, stage 2; f) Social/cultural, stage 3; g) Human Health, stage 2; h) Human Health, stage 3. Blue=US/CA scientists; Red=AU scientists; Green=US/CA managers; and Purple=AU managers. Each species is a spoke of the graph, with corresponding species directly opposite each other via the line through centre: CS=*Caulerpa scalpelliformis*, UAlg='Unknown Algae', PV=*Pterois vol*itans, UF='Unknown Fish', BO=*Bonamia ostreae*, UP='Unknown Parasite', MR=*Maoricolpus roseus*, UG='Unknown Gastropod', CI=*Ciona intestinalis*, and UAsc='Unknown Ascidian'.

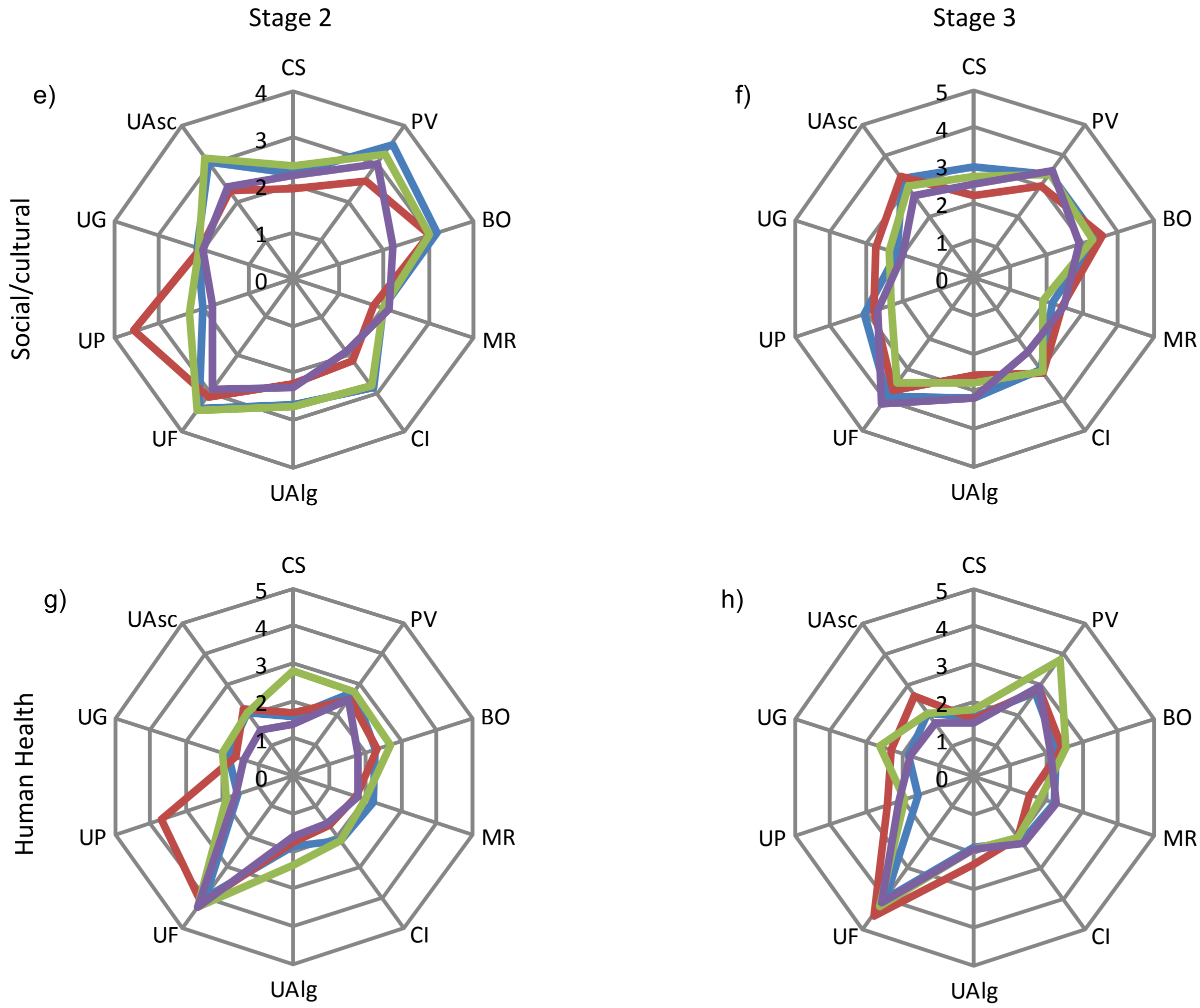

Figure 4.4 cont. Mean consequence assessments of each species for each workshop group at stages two (2) and three (3) and for each core value, with: a) Environmental, stage 2; b) Environmental, stage 3; c) Economic, stage 2; d) Economic, stage 3; (overleaf) e) Social/cultural, stage 2; f) Social/cultural, stage 3; g) Human Health, stage 2; h) Human Health, stage 3. Blue=US/CA scientists; Red=AU scientists; Green=US/CA managers; and Purple=AU managers. Each species is a spoke of the graph, with corresponding species directly opposite each other via the line through centre: CS=*Caulerpa scalpelliformis*, UAlg='Unknown Algae', PV=*Pterois volitans*, UF='Unknown Fish', BO=*Bonamia ostreae*, UP='Unknown Parasite', MR=*Maoricolpus roseus*, UG='Unknown Gastropod', CI=*Ciona intestinalis*, and UAsc='Unknown Ascidian'.

Distribution

There was a statistically significant effect of distribution for the "Unknown Gastropod" and *Maoricolpus roseus* for the second (human health) and third (all values) stage (Table 4.2). Where *M. roseus* is statistically different (third stage of environmental, economic and social/cultural values), AU participants consistently ranked it greater than did US/CA participants. Where Unknown Gastropod is statistically different (second stage of human health and third stage of environmental and human health values), US/CA participants ranked it greater than did AU participants.

Table 4.2. Differences in consequence magnitude pooled for each country. Bold font indicates the greater value (statistically significantly different at p<0.05) between the two countries.

Core Value (Stage)	Species	US/CA mean	AU mean
Environmental (3)	Unknown Gastropod	**3.24**	2.48
	Maoricolpus roseus	2.32	**3**
Economic (3)	*M. roseus*	1.96	**2.57**
Social/cultural (3)	*M. roseus*	2.04	**2.48**
Human health (2)	Unknown Gastropod	**1.96**	1.52
Human health (3)	Unknown Gastropod	**2.45**	1.95

Effects of species name and core value bias

Though results indicated no consistent difference by workshop group or stage of the assessment process for species name or core value, when there was a significant difference, participants consistently ranked Unknown Algae (species name bias) and *B. ostreae* (core value bias) greater than their respective counterparts (*C. scalpelliformis* and Unknown Parasite, respectively; Table 4.3).

Table 4.3. The effects of species name and core value are summarized by workshop group. CS=*C. scalpelliformis*; UAlg=Unknown Algae; BO=*B. ostreae*; UP=Unknown Parasite. Bold font indicates the great mean consequence magnitude; *indicates not significant at α=0.05. Sc=scientist; Mg=manager; US/CA = United States/Canada; AU= Australia.

Group (Stage)	Species name effect (p-value)	CS mean consequence	UAlg mean consequence	Core value effect (p-value)	BO mean consequence (econ)	UP mean (env)
US/CA Sc (2)	NS*	-	-	P=0.018	**4.30 +/- 0.21**	3.43 +/-0.20
US/CA Sc (3)	0.005	3.92 +/-0.10	**4.29 +/-0.11**	NS*	-	-
AU Sc (2)	NS*	-	-	P=0.006	**3.92 +/-0.18**	3.00 +/-0.28
AU Sc (3)	P=0.006	3.36 +/-0.20	**4.00 +/-0.00**	NS*	-	-
US/CA Mg (2)	NS*	-	-			
US/CA Mg (3)	NS*	-	-	P=0.006	**3.75 +/-0.13**	3.08 +/-0.29
AU Mg (2)	P=-.031	3.50+/-0.22	**3.70 +/-0.15**	NS*	-	-
AU Mg (3)	P=0.005	3.79+/-0.15	**4.21 +/-0.11**	NS*	-	-

Relationship between consequence and uncertainty assessment

For all workshop groups, there was an inverse (negative) correlation between uncertainty and consequence: high uncertainty ratings were associated with lower consequence ratings and conversely, higher consequence ratings were associated with lower uncertainty ratings. Spearman's correlation tests were significant for consequence and uncertainty (for US/CA scientists, *r*=-0.46; for AU scientists, *r*=-0.42; for US/CA managers, *r*=-0.56; for AU managers, *r*=-0.44) (Figure 4.5a-d).

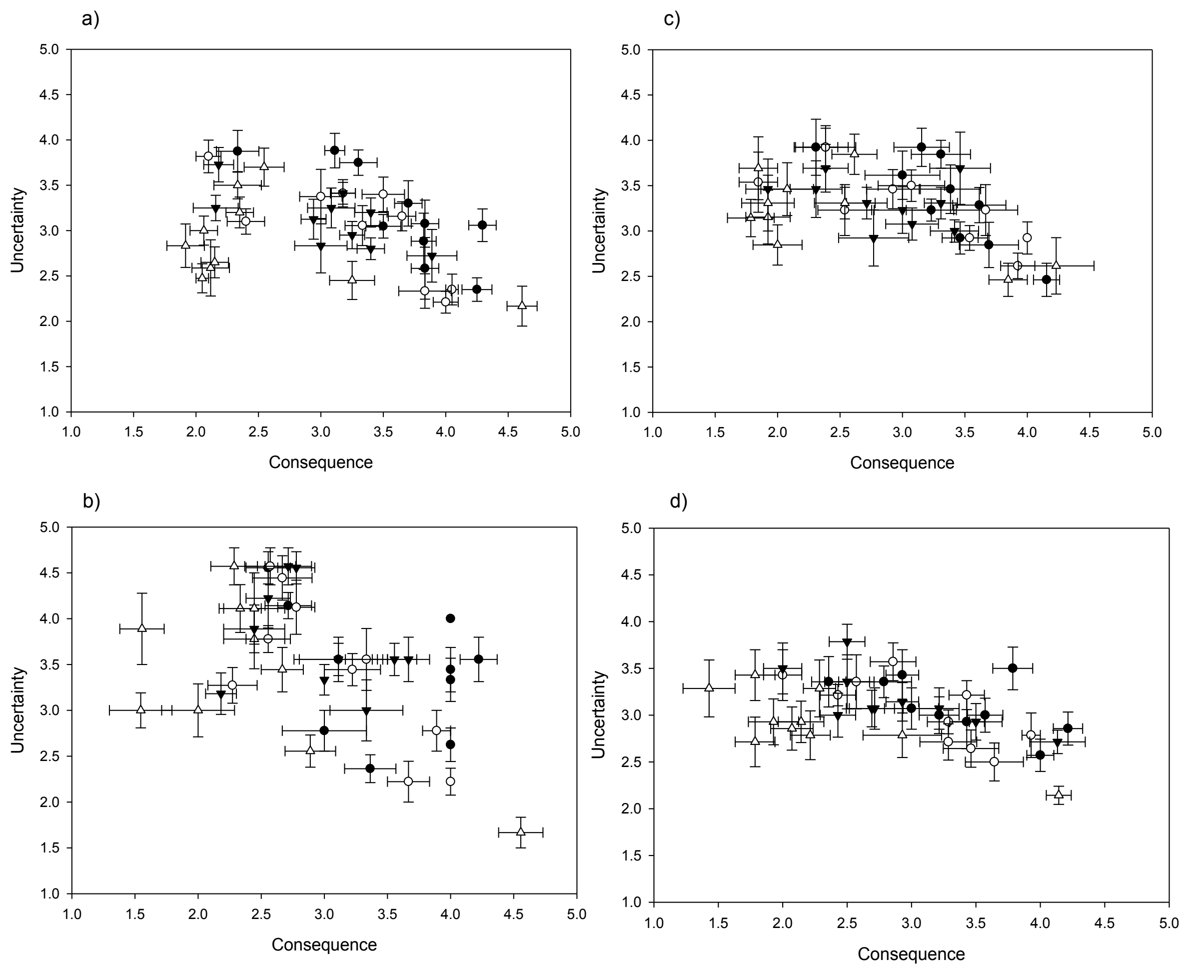

Figure 4.5. The level of consequence and uncertainty for the range of core values after group discussion, with standard error (SE) bars (horizontal SE bars=consequence; vertical SE bars=uncertainty) for: a) US/CA Scientists (MBIC); b) AU Scientists (AMSA); c) US/CA Managers (ICAIS); and d) AU Managers (NIMPCG). ●=environmental consequence and uncertainty, **O** =economic consequence and uncertainty; ▼=social and cultural consequence and uncertainty; **Δ** =human health consequence and uncertainty.

Discussion

This study broadly focused on determining how science and management experts made decisions under uncertainty using consequence assessment for 10 ANS (5 real and 5 theoretical). Specifically, I determined whether the experts assumed impact (or not); if any heuristics or biases such as the species' geographic origin, species name or core value, affected the experts' assessments; and if any of these varied by the expert's origin (North America or Australia).

Literature on the relationships between uncertainty and impact or consequence assessment has primarily focused on the presence and effects of external uncertainty. For example, risk perception research has asked individuals to hypothetically rate the risk of threats such as climate change, given a specified level of uncertainty surrounding the event (e.g., Lazo et al. 2000, IPCC 2001). Alternatively, environmental impact assessment research has asked the scientist to estimate the uncertainty surrounding their estimate of impact to marine ecosystems (Halpern et al. 2007). These studies assume the perceived risk or consequence is the primary driver, with uncertainty as an external and secondary descriptor, or by-product of this initial assessment. Despite significant effort to understand the relationship between uncertainty and risk perception and assessment, previous research has not reached a consensus on the issue (Wiedemann et al. 2006). Applied risk assessment methods may ask experts to assess the uncertainty around their impact or consequence rating, but often do not examine how their uncertainty influenced their initial rating. This is a fundamental difference given the former is a conscious action while the second is the result of an unconscious influence.

Decision Theory provides general (and often normative) guidance on the influence of uncertainty on individual DMURI. The HSM and risk perception research offer descriptive theories and empirical findings of DMURI. However, little research has been done on the influence of an expert's uncertainty on their conservation (Donlan et al. 2010), or biosecurity-related decisions. The potential use of heuristic and systematic approaches (as described by HSM) in situations of uncertainty is also largely unknown (Trumbo and McComas 2003). Experts are not exempt from the influence of these heuristics, particularly in situations of scarce information and high uncertainty (Sjoberg 2002). Hence, understanding the potential effects of these factors on the outcomes of risk assessment is a priority.

To this end, this study offers several preliminary conclusions on the specific heuristics and biases that result from a combination of SIFs and species' attributes. The four main trends from this research were:

- the presence of a species in an expert's region may increase the expert's perception of impact;
- a species name may exert influence on assessment;
- a core value bias exists that influences individual expert perceptions; and

- experts tend to assume (and assign) lower consequence when faced with knowledge gaps and other forms of uncertainty.

Differences between groups: by group and by region of origin

The differences between workshop groups in the second stage assessment consisted of differences in Australian managers' ratings. However, these differences disappeared in the third stage, perhaps due to the tendency for consensus in Delphic discussion (Webler et al. 1991, Patton 2002). Geographical-based differences also occurred. For example, Australian groups rated *M. roseus* higher than North American groups for environmental, economic and social/cultural values and North American groups rated Unknown Gastropod higher than Australian groups for environmental and human health values. This difference aligns with expectations given the presence of *M. roseus* as a nationally recognized, but not internationally significant, ANS in Australia. It is introduced to Australia but not North America. Australian experts were likely familiar with *M. roseus*, but the absence of a global invasion history would suggest a lack of awareness in North America.

Conversely, the fact sheets described the Unknown Gastropod as introduced to North America but not Australia, which also suggests increased proximity of the species increased perceived consequence. This supports past research that found decreased distance from a hazard was associated with increased awareness and perception of the risk (Lerner et al. 2003, Smith 2008, Kuhar et al. 2009). The availability heuristic may also play a role in this result. For example, a local ANS may get more attention in the local media, research laboratories or simple discussion, which may increase an expert's perception of the impact. In addition, Australia has fewer harmful gastropod introductions, which may lead to a greater effect of recognition heuristic by North American participants. This may occur if North American experts associate frequency of identified cases of specific gastropod species impact in their region with the general impact of gastropods (an association known as surrogate correlation; Goldstein and Gigerenzer 2002). Whether or not the former will be an accurate mediator for actual impact of gastropods (and hence the correct prediction for a novel gastropod species) will depend on the ecological correlation (the correlation between the true impact of gastropods and frequency of harmful local gastropod introductions) (Goldstein and Gigerenzer 2002). Given that some geographic areas have been identified as potentially more susceptible to invasion (Ruiz et al. 2000), using local case studies to predict future impacts by the same taxa may potentially be a valid heuristic approach.

Species name influence

I found evidence for species name bias in the third stage of assessment for both scientist groups, with no clear trend by stage otherwise (AU managers perceived a significantly greater consequence of the Unknown Algae for both stages). However, all groups consistently had relatively higher consequence ratings for Unknown Algae (versus *C. scalpelliformis)*. This was unexpected given the well-known impacts of species in the *Caulerpa* genus (e.g., *Caulerpa taxifolia* and *Caulerpa racemosa*). Analysis of the discussion may explain the lower consequence rating for *C. scalpelliformis*: participants clarified that *C. scalpelliformis* wasn't *C. taxifolia* and that the effects of *C. taxifolia* weren't as great as some sources would indicate. In addition, AU managers mentioned the native status of *C. scalpelliformis* in Australia, as well as a perception that *Caulerpa* species only established in heavily degraded areas, which were already heavily impacted and therefore of less concern.

Core value bias

Again, both scientist groups displayed a core value bias in the second stage, though no consistent pattern otherwise. However, where a significant difference occurred, participants ranked the economic effect of *B. ostreae* significantly higher than the environmental consequence of the Unknown Parasite. The removal of bias at the third stage for the scientist groups may also have been due to discussion. Several groups mentioned (and considered important) the fact that *B. ostreae* affected a nonindigenous oyster, while reiterating the ecological importance of the species effected by the Unknown Parasite (a seastar). The US/CA manager group showed this bias in a reverse direction: no difference at the second assessment, while the economic consequence of *B. ostreae* was rated higher than the environmental consequence of the unknown parasite. The (economic-focused) discussion may also have played a role: there was discussion of the unimportance of the seastar, particularly economically.

These results indicate a present, but rather limited, role of bias for the assessment of these particular species. This suggests that monitoring discussion and reviewing the rationale for decisions remains important (Patton 2002). However, other heuristics not specifically analysed but identified in other research on DMURI may warrant investigation in a conservation or biosecurity setting. For example, availability (recall bias) may be responsible for the approach by some participants of associating the Unknown Ascidian with high-profile ascidians (e.g., *Styela clava* and *Didemnum* spp.)

and assigning it an impact. The anchoring heuristic[10], while not solidly assessed (it was impossible to track changes on the poster, by individual), may have occurred via 'leaders' in the groups that had more experience and influenced other participants. Future work should include analysis of these alternative heuristics. It should be noted that the aim of this component was identification and understanding of these heuristics – their identification does not imply they are wrong or necessarily 'bad'; heuristics can provide an accurate judgmental shortcut in cases of uncertainty. This is particularly true if heuristics appropriate to the environment with demonstrated accuracy are applied (Hogarth and Karelaia 2007). For example, if crustacean species have repeatedly shown a high impact on community biodiversity, using the representativeness heuristic to make a decision under uncertainty for a new crustacean may improve the biosecurity management outcomes.

A discussion of power

The analyses for effect of group, region, species name, and core value were likely hampered by low power. The small sample size (12-21 participants in each group; 27-34 when grouped by origin or expert type) and small effect size (e.g., differences in means of < 1) would have made detecting a difference difficult. As G*Power3 does not address nonparametric tests, I tested one set of data from a Kruskal Wallis test using the equivalent parametric ANOVA. With N=30, a small effect size (f=0.15) and four groups, the associated power was 0.08. That is, this test had an 8% probability of detecting a result. Given that nonparametric tests are even less powerful than their parametric equivalent, the justification for drawing conclusions based solely on insignificant results is not justified. As such, I use descriptive along with inferential statistics to optimize the value of conclusions and recommendations. While the sample size was limited by logistics, future work should include better design that maximizes power.

The hindsight approach as a potential heuristic

In a biosecurity context, several management alternatives exist to address a species of unknown probability and consequence (conditions of ignorance). One can assume no impact and do nothing, or assume an impact and take steps to prevent or manage the incursion. These options have four potential outcomes:

1) no impact is assumed (nothing done) and species has no impact;

[10] The tendency to be influenced by initial estimates, making estimates based on others' judgments, or even deferring judgment to others deemed to have more authority or expertise (Tversky and Kahneman 1974).

2) impact is assumed (steps taken) and species has no impact;

3) impact is assumed (steps taken) and species has impact; and

4) no impact is assumed (nothing done) and species has impact.

Assuming a species having an impact is a ‘worst’ case scenario, following the Maximin Principle (as described by Decision Theory), requires choosing the alternative that produces the better of these two, such as taking steps to mitigate the impact. Precaution suggests a similar approach. In conditions with some indication of harm that is serious or irreversible, yet uncertain (conditions common to many conservation and biosecurity decisions), precaution advocates erring on the side of protecting the resource at risk (Lauterpacht International Law Centre 2000).

The outcomes of this study showed a negative correlation between consequence and uncertainty for all groups, with a slightly stronger effect for US/CA-based groups. Based on these outcomes, participants appear to make decisions in violation of the normative Maximin Principle and, in particular, precaution. Yet in the initial survey, 79% felt it more important to avoid making Type II errors and 87% viewed precaution as a necessary component of a risk assessment. Other studies have found similar patterns. For example, Wilson et al. (2006) surveyed senior policy officials in Canada to assess their use of precaution in light of a recently-developed national framework for the use of precaution. The survey showed that policy officials applied the precautionary framework rarely or not at all, despite a general support for precaution (Wilson et al. 2006).

As an alternative to the normative Maximin Principle, descriptive heuristics may offer alternative explanations for these outcomes. The Heuristic-Sufficiency Model (HSM) posits that individuals will process information one of two ways: heuristically or systematically. This is based in part on information sufficiency (the individual’s perceived adequacy of the available knowledge), which is an inverse proxy for uncertainty (Trumbo 2002). A perceived insufficiency or absence of desired information can lead to feelings of information uncertainty, leading to interpretation according to heuristics (Lion et al. 2002, Kahlor et al. 2003). In this exercise, participants often felt they did not have enough information to make a decision, which, according to the HSM, suggests they were more likely to evaluate species via heuristic methods (Chen and Chaiken 1999, Trumbo 2002). Conversely, when they did have sufficient information (i.e., when scientific literature was available), they could (and likely did, based on motivation) make a systemic evaluation of the consequences. Based on these outcomes, the question becomes, what heuristic(s) did the experts use to assign consequence when they were forced to make decisions in situations of high uncertainty, such as absence of information?

Sjoberg (2002) suggested that the norms in specialist fields often run deep and may become ingrained and create a filter through which the individual perceives risk, consciously, or unconsciously (the use of heuristics is often unconscious; Tversky and Kahneman 1981). One of the earliest and most important norms learned and incorporated by scientists is null hypothesis significance testing and the application of a significance criterion (α), which assume no difference (e.g., impact) unless the *p*-value (which is the false positive rate, or rate of Type I errors) is below a certain significance level (conventionally set at 0.05). Otherwise, the experimenter assumes "no difference." In impact assessment, this correlates to assuming no impact of an activity or species unless the probability of a Type I error rate is sufficiently low. This focus on Type I error rates is characteristic of the "innocent until proven guilty" approach (no effect without sufficient (scientific) evidence), which may have deleterious implications in biosecurity (see Chapter 5).

This "innocent until proven guilty" assumption, or "hindsight heuristic" not only appears unconscious (given survey responses to Type II error avoidance and precaution use), but also fits additional characteristics identified by the literature as likely to contribute to a heuristic's use. These include a high degree of relatedness and relevance (ANS scientists often study impact), frequent historic use (experimenters almost universally apply the significance testing methods described above), and ease of recall (not a difficult tenet to remember) (Chen et al. 1999). The use of this heuristic may explain the lower consequence assessments when the participants were uncertain and higher evaluation when they felt they had sufficient information (e.g., scientific literature). That is, it suggests that both scientists and managers use a hindsight approach when making decisions under uncertainty (Figure 4.6). Given Australia's increased application of precaution, Australian experts may apply this heuristic slightly less strongly; however, the difference, if any, appeared slight and suggests they, too, tended to assume less impact when uncertain.

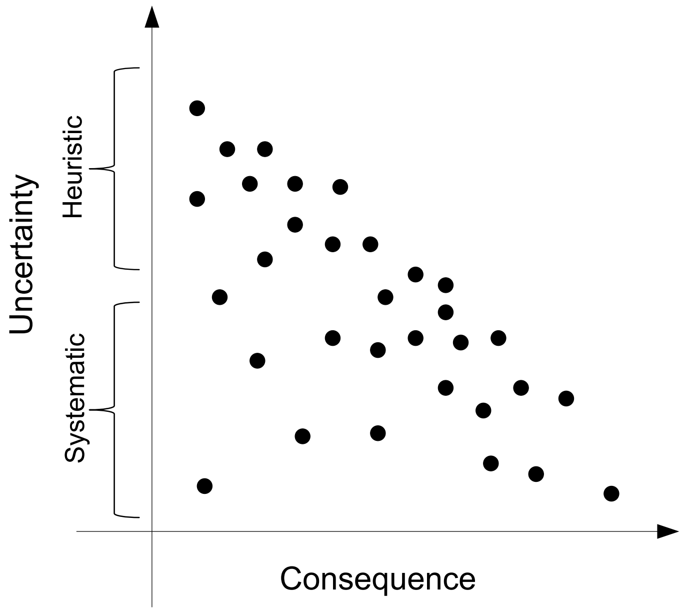

Figure 4.6. Proposed model of decision making under uncertainty.

These outcomes have implications for the identification and management of uncertainty within risk assessment. The scarcity of ANS impact information and associated high uncertainty may potentially lead to decisions based on heuristics (which may or may not be appropriate). In this study, the most common heuristic was an assumption of innocence without evidence. In an impact assessment context, this approach may have several consequences. Primary among these is one common in statistical decision theory: the trade-off between Type I (falsely assigning an impact) and Type II (missing an impact) error rates. As the precaution approach assumes an impact, it will tend to have higher rates of Type I errors; as the hindsight approach assumes no impact, it will tend to have higher rates of Type II errors. Impact assessment may be an inappropriate field for the traditional focus on avoiding Type I errors due to the higher potential cost of Type II over Type I errors (Page 1978, Fairweather 1991). An additional concern given this relationship is the additional weight sometimes given to “certain” judgments (e.g., Halpern et al. 2007). Not only will an impact potentially be rated higher given feelings of certainty, its relative importance may again increase if assessors perceive certain effects to have more consequence.

Thus, the consistent use of the hindsight approach (in which Type II errors are more common than Type I errors) may create a bias against environmental conservation and sustainable management (Mapstone 1995). This may be particularly true for ANS impact assessment. ANS impacts often occur over a variety of spatial and temporal scales (including lag effects; Crooks and Soule 1999, Ricciardi 2003) in areas with other anthropogenic impacts, making the detection and isolation of impacts difficult and potentially leading to increased Type II error rates. Given these arguments, some experts argue for the use of precaution and its implementation via assuming guilty until proven innocent and subsequent minimization of Type II errors (e.g., Buhl-Mortensen 1996, Hewitt et al. 2006). While acknowledging the importance of epistemological rationality and the associated focus on Type I errors in pure science, Buhl-Mortensen (1996) argues that applied science must also make decisions via considerations such as conflicting attitudes toward risk, and ethical and legal obligations. Precedents for using precaution in research and development occur in the fields of new pesticides, food or drug products (Belsky 1984, Schrecker 1984). Peterman (1990b) argues that the management of natural resources should be no different.

These outcomes represent a preliminary attempt to understand decision making in a consequence assessment context, for a range of core values and including both scientists and resource managers in two different contexts (Australia and the US/CA). Due to data scarcity and the associated increase in expert judgment as a decision-making tool, identifying how cognitive processes affect the outcomes of risk assessment and managing them accordingly remains an important goal. While this study focused on ANS, the outcomes may be applied to the management of other conservation threats. For example, the '*a priori'* influence of uncertainty suggests the usual approach of indicating '*post hoc'* the uncertainty associated with a consequence estimate may not be sufficient to account for the effects of cognitive processes on the initial assessment, particularly if precaution is desired. Below, I discuss some potential strategies to address the influence of uncertainty and related heuristic processes in a risk or consequence assessment context, as well as integrating precaution in a transparent and flexible manner.

Recommendations

As these recommendations provide guidance for both research to inform management, as well as risk and management decisions, they can be applied together for synergistic effects. However, if not appropriate or feasible for a particular situation, they may be used independently.

- **Use the 'hindsight heuristic' in expert assessments to develop research and management priorities.** The scatter plots of consequence and uncertainty suggest potential conservation or biosecurity management strategies if divided into four quadrants: (1) high impact with low uncertainty; (2) high impact with high uncertainty; (3) low impact with high uncertainty; and (4) low impact with low uncertainty (Figures 4.7 and 4.8, with application to species reviewed in assessment provided in Table 4.4).

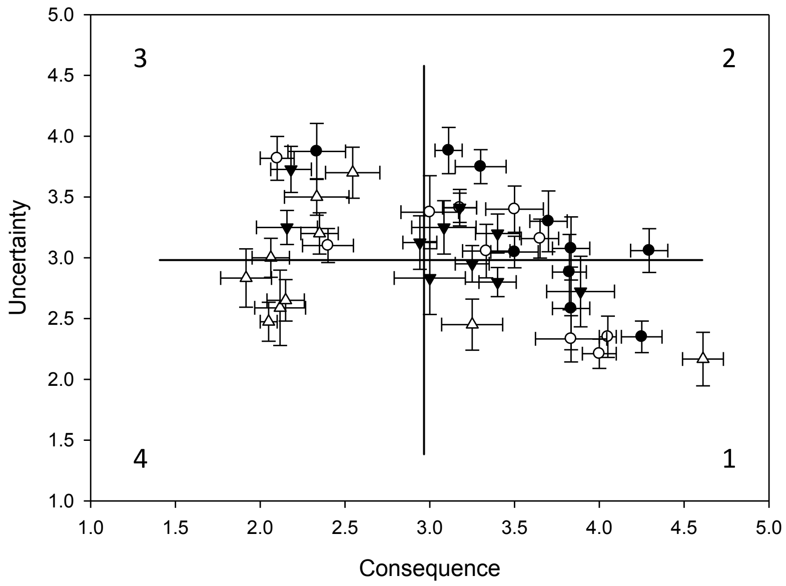

Figure 4.7. Consequence and uncertainty plot for the range of core values assessed, with standard error (SE) bars (horizontal SE bars=consequence; vertical SE bars=uncertainty). ●=environmental consequence and uncertainty, **O**=economic consequence and uncertainty; ▼=social and cultural consequence and uncertainty; **Δ**=human health consequence and uncertainty. The four quadrants represent: (1) high impact with low uncertainty; (2) high impact with high uncertainty; (3) low impact with high uncertainty; and (4) low impact with low uncertainty.

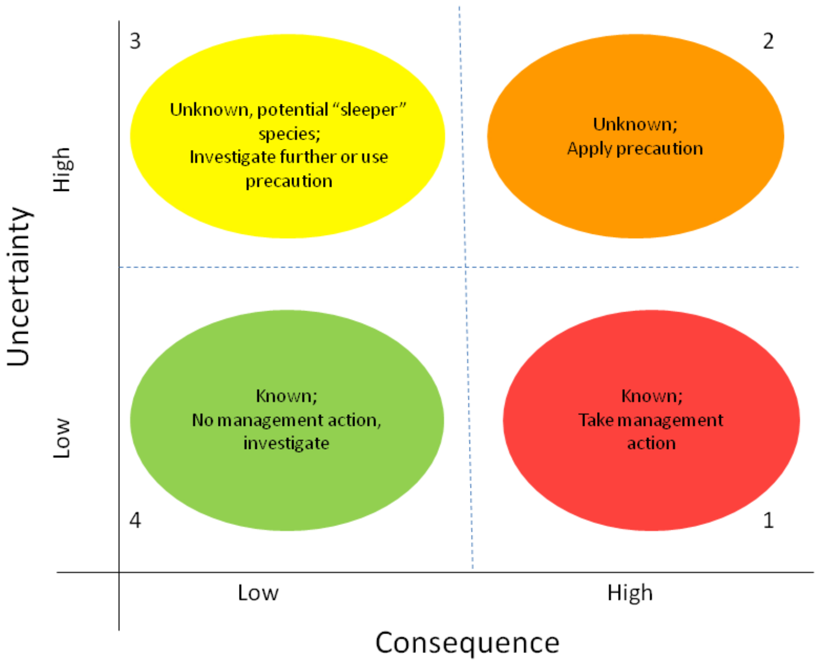

Figure 4.8. Management model for the division of consequence versus uncertainty divided into four quadrants for managing ANS (or other conservation-based entities). The four quadrants represent: (1) high impact with low uncertainty; (2) high impact with high uncertainty; (3) low impact with high uncertainty; and (4) low impact with low uncertainty.

Two of these provide clear direction for management; species (or other entities) in quadrant 1 represent a high management priority (as suggested, e.g., by Donlan et al. 2010) while those in quadrant 4 represent a low priority (Figure 4.8). Species in quadrants 2 and 3, however, present a challenge. Those in quadrant 2 are similar to those in quadrant 1: they will likely have a large impact, but uncertainty is high and the exact magnitude unknown. The results of this study suggest treating these species with precaution, that is, similarly to those species in quadrant 1 due to the evidence that assessors are reluctant to mark a species with high consequence and uncertainty as high magnitude (in addition to remaining a research priority; Donlan et al. 2010). If a species with high associated uncertainty is also marked high magnitude, it is highly likely to be justified by direct evidence or other justification. The challenge of managing species in quadrant 3 also results from the trend of assuming innocence (low impact) without evidence; whether or not that species actually has a low impact relates less to actual impact potential but more to the hindsight heuristic. Despite this, WTO requirements (i.e., a scientifically-based risk assessment) do not justify marking a species "high risk" without any supporting evidence. Any decisions or policy based on such a decision may not hold up to outside scrutiny (e.g., a WTO challenge[11]). One option is to mark the consequence as 'low' and keep the species within the scope of the risk assessment (an approach also recommended by survey outcomes), so that any new evidence can be easily incorporated to obtain a new risk estimate. This option provides a low level of precaution while unlikely to incur increased management costs or efforts (e.g., Possingham et al. 2002), given the management actions for high consequence species will likely include those necessary for low consequence species. If time and budget resources permit, additional research should target these 'sleeper' species.

Table 4.4. Management categories for the 10 ANS considered in the assessment, averaged across all workshops. "Max" represents the largest score from the 4 core values.

Species	Environment	Economic	Social/Cultural	Human Health	Max
Caulerpa scalpelliformis	2	3	3	3	2
Unknown Algae	1	3	3	3	1
Pterois volitans	1	1	2	1	1
Unknown Fish	2	2	2	1	1
Bonamia ostreae	2	1	2	3	1
Unknown Parasite	1	3	3	3	1
Maoricolpus roseus	3	3	3	3	3
Unknown Gastropod	3	3	3	3	3
Ciona intestinalis	1	1	3	4	1
Unknown Ascidian	2	1	3	3	1

- **Provide the option to indicate uncertainty and apply precaution in a transparent way.** Risk assessments often use a risk matrix (which is linguistically qualitative) to combine the likelihood and consequence components. Generally, the value for each component is plotted in the matrix, with the resulting risk estimate based on where they intersect, leaving uncertainty unaccounted for. Uncertainty could be incorporated in several ways, depending on the desired level of precaution. One option (Figure 4.9; solid lined arrow) would use the average consequence rating, with the standard error arround the estimate representing the uncertainty. Another option (Figure 4.9; dashed lined arrow) would represent consequence as the full range of assessed consequence. For example, if the lowest rating was 'negligible' and the highest was 'high', extending the consequence into these categories would provide

[11] While it is possible a WTO member may challenge on the basis of SPS Agreement Article 2 (i.e., "Members shall ensure that any sanitary or phytosanitary measure...is based on scientific principles and is not maintained without sufficient scientific evidence"; WTO 1995), Members are free to sovereignly establish their own acceptable level of risk, as long as it is consistent across trade-related decisions (Campbell et al. 2009).

a risk range instead of just a single outcome. For example, this might be moderate to high, rather than moderate. Presenting risk as a range has also been used in other risk assessments, for example, to show differences in risk between individuals or regions (Campbell 2008) in a freshwater biosecurity assessment. While this approach still places uncertainty on the 'outside' of the estimate, it allows an assessor to counteract the hindsight heuristic and incorporate precaution (if that is desired). Thus, with a range of risk estimates, an assessor may choose to apply more (in the form of choosing the higher estimate) or less (lower estimate) precaution.

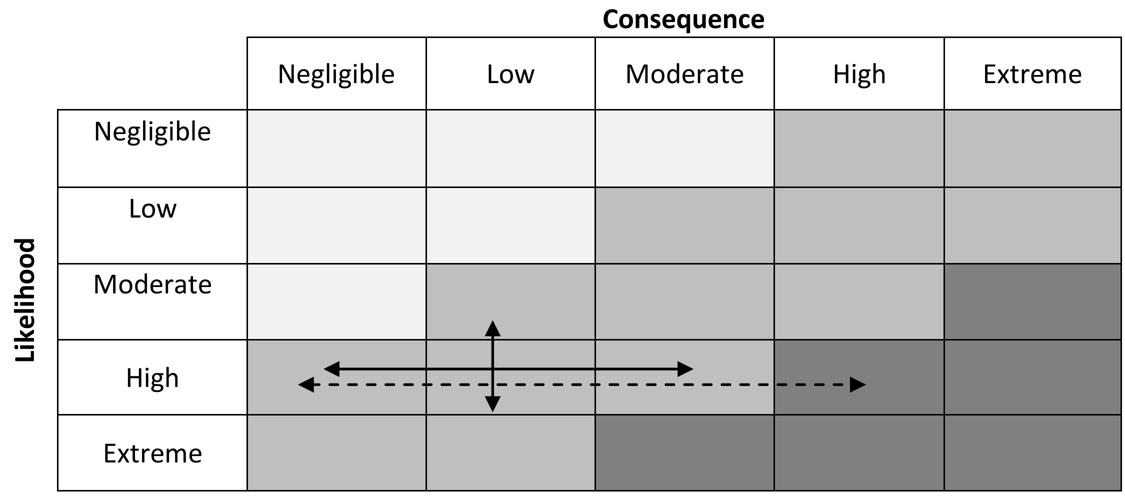

Figure 4.9. Risk matrix with arrows representing the range of uncertainty. Solid arrows represent an average with standard error, while the dotted arrows represent a range approach. Shades of grey represent low, moderate and high risk.

Conclusion

Given the magnitude of ANS impacts on conservation goals, understanding actions that can mitigate these impacts despite uncertainty is imperative (Donlan et al. 2010). The results of this chapter demonstrate that uncertainty need not be a paralysing influence on risk assessment and management decisions. Use of the methods described above provides a means to account for uncertainty, as well as integrate the controversial and confusing concept of precaution into the consequence assessment in a transparent and objective manner that does not necessarily need "precautionary" as a descriptor. This moves the discussion of precaution from a subjective component of risk assessment (where it runs up against barriers in the WTO risk assessment requirements) and into policy decisions (e.g., risk management and setting of the acceptable level of risk). The use of precaution is established in this area, given the national sovereignty principles provided by the WTO to nations for setting polices such as the acceptable level of risk.

The assumption that all nonindigenous species should be treated as a potential threat has been challenged by some (e.g., Brown and Sax 2004, Davis et al. 2011, but see Simberloff et al. 2011, Alyokhin 2011, Lerdau and Wickham 2011), who argue that too often, we judge these species negatively and attempt to eradicate them *ipso facto*, without any solid, supporting evidence. While costly control methods should certainly be assessed and prioritized, resulting in some nonindigenous species inevitably left alone in favour of managing the more invasive offenders (Simberloff et al. 2011), the recommendations provided herein do not focus on the eradication process, addressing instead prevention and early control efforts. In this latter pursuit, data is unfortunately not the solution, but often the problem – sufficient data just does not exist for many species and is not feasibly attainable with policy and management timeframes (Lerdau and Wickham 2011). As such, these outcomes and recommendations do not represent a baseless condemnation of nonindigenous species. Rather, they serve as tools to better complete consequence assessments that address uncertainty and (if desired) incorporate precaution in a manner that reflects the demonstrated value of prevention and facilitates more comprehensive and expedient consequence and risk assessments. The flow-on effect of this process is the improved ability of biosecurity and other management agencies to protect threatened marine resources.

CHAPTER 5. EVIDENCE OF IMPACT: LOW STATISTICAL POWER LEADS TO FALSE CERTAINTY OF NO IMPACT FOR NONINDIGENOUS ALGAL AND CRUSTACEAN SPECIES

Manuscript in preparation:

Dahlstrom, A. and C. L. Hewitt. 2011. Evidence of impact: low statistical power leads to false certainty of no impact for nonindigenous species. Nature (in prep).

Dahlstrom, A (80%), Hewitt, CL (20%)

- CL Hewitt contributed to the idea, its formalization and development, and assisted with refinement and presentation.

Introduction

There is a general recognition that biological invasions have the potential to cause significant impacts on core values (National Research Council 1996b, Carlton 2001, Hewitt 2003a, Hewitt and Campbell 2007, Campbell 2008). Despite this recognition, large numbers of scientists and managers repeatedly describe species as having no impact, when in reality most of these assessments are due to a lack of evidence (Carlton 2002).

When estimating a species' or vector's consequence and subsequent risk in a biosecurity context, decision makers apply a systematic or heuristic approach (or a combination of both; see Chapters 3 and 4). While the role of and associated influences on heuristic processing in risk assessments have been increasingly recognized due to the uncertainty often present (Fairbrother and Bennett 1999), most biosecurity risk assessment instruments still espouse a primary reliance on systematic processing based on available science-based knowledge (Dahlstrom et al. 2011).

As discussed in Chapter 4, most fields, including biology and ecology, use significance testing, or more formally, null hypothesis significance testing (NHST), as the traditional and generally-accepted procedure for drawing conclusions based on experimental results, in order to augment existing scientific knowledge (Weinberg et al. 1986, Rosnow and Rosenthal 1989, Anderson et al. 2000, Altman 2004, Fidler et al. 2004, Hobbs and Hilborn 2006, Nakagawa and Cuthill 2007). NHST consists of statistically evaluating whether a set of results differs from a pre-identified null hypothesis through statistical testing. This test (often defined as "no difference") informs the decision whether to reject or fail to reject the null hypothesis based on the probability (p-value) that the findings are unlikely to be within the population of the control. The p-value represents the probability, assuming the null hypothesis is true, of observing data at least as extreme as that in the study, and is compared to a pre-determined acceptable level (e.g., 0.05) (Lehmann and Romano 2005).

The use and creation of null hypotheses has been the focus of much discussion. For example, Karl Popper believed that falsification of theories (e.g., via an attempt to reject the null hypothesis) led to a steady advancement of scientific knowledge (Popper 1959, Cohen 1994). In contrast, Thomas Kuhn believed scientific advances occurred in a "normal science" context, in which individuals worked within and generally in support of the existing paradigm, until sufficient evidence against the paradigm caused a revolutionary shift (Kuhn 1996, Graham and Dayton 2002). These paradigms include the status quo methods by which data are collected, analysed and compared (Graham and Dayton 2002), by which definition, it could be argued, the assumptions and methods within NHST itself may form just such a paradigm.

While aiding the justifiable attempt to reduce uncertainties in our knowledge of the world around us, some applications of the NHST paradigm may inadvertently create 'false certainty'. In the case of biosecurity, false certainty can arise when 'statistically non-significant' results are used to attribute a finding of 'no impact' to a species or vector. This problem becomes particularly important in cases of low power (i.e., a low probability of detecting an effect). False certainty may not only cause an inaccurate understanding of the world, but may also cause actual harm in some circumstances. In an impact assessment context, it can obscure effects on a native species, community or ecosystem due to insufficient sample or effect size or inappropriate experimental design, leading to low power and Type II errors. Additional discussion of NHST criticisms contributing to false certainty are presented below.

Criticisms

Significance testing has been criticised for as long as *p*-values have routinely been reported (Stewart-Oaten 1996, Fidler et al. 2006), particularly in such areas as impact assessment (Schmitt et al. 1996). Many fields (e.g., statistics, economics, business and engineering) have placed significance testing as one tool in a larger tool box, while biology and ecology tend to retain the singular focus on *p*-values (Yoccoz 1991, Stewart-Oaten 1996). Criticisms fall into several different (though related) areas discussed below:

1) Statistical significance does not equal biological significance

    Many investigators judge the usefulness and/or validity of a study on whether or not the p-value is above or below 0.05, due to the belief that this threshold is a reliable, convenient and rigorous standard sufficient to prevent the false acceptance of results as significant (Yoccoz 1991). Much criticism of NHST is based on this strong dependence or emphasis given to the outcome of the statistical test, particularly the attribution (or not) of biological or ecological significance to something that is statistically (non)significant (Weinberg et al. 1986, Yoccoz 1991, Mapstone 1995). Rejection of the null hypothesis may not be relevant to any real causal difference, while a failure to reject the null hypothesis may miss a causal difference if the power of detection is insufficient (Rosnow and Rosenthal 1989, Weiss 1999, Ziliak and McCloskey 2008). For example, in a study of the bivalve *Gemma gemma* and polychaete *Clymenella torquata*, Weinberg (1986) found that the demographic importance of statistically significant interactions varied widely. Some individuals even apply this attribution of significance along a scale, where results with $p<0.0001$ are ranked as 'more' significant (i.e., important) than those with a p-value of 0.049

(Ziliak and McCloskey 2008), despite a lack of biological, economic, social, cultural, etc., relationship or meaning to either α[12] or β[13] (Mapstone 1995).

2) The dichotomous 'reject/fail to reject' at α=0.05 is arbitrary

Significance testing ignores an assessment of the likelihood of the null and alternative hypotheses, favouring simple acceptance or rejection despite the lack of any ontological validity in this approach (Nakagawa and Cuthill 2007, Stephens et al. 2007). The dichotomous choice based on a fixed significance level was initially applied to agriculture experiments (Fisher was an agronomist; Cohen 1990), which is a situation far different from the nuanced research questions in many modern studies. As such, some experts in ecology and conservation science have questioned this dichotomous decision procedure (e.g., Fidler et al. 2006, Stephens et al. 2007). Some argue for the choice of a significance level based on the time, place and investigator. For example, Enrico Fermi considered a *p* of 0.10 the definition of a "miracle" (Polanyi 1962), and other published works view 0.15, 0.0325 or 0.07 as "firm evidence" (Stigler 2008). Thus, instead, the value of results (based on the *p*-value) falls along a significance continuum – i.e., as put by Rosnow and Rosenthal (1989), "surely, God loves the .06 as much as the .05".

3) Traditional significance testing ignores effect size

Some (e.g., Nakagawa and Cuthill 2007, Stephens et al. 2007) argue that significance testing actually pursues a somewhat unusual goal, such as to determine a difference between two populations. However, this can be described as a rather obvious fact, given that with a large enough sample size, two different treatments will always show a difference (Johnson 1999). This difference, known as the effect size (ES), provides an estimate of the magnitude and direction of an effect (Nakagawa and Cuthill 2007), the knowledge of which will lead to greater understanding of the issue (Gaines and Denny 1993, Schmitt et al. 1996, Nakagawa and Cuthill 2007), particularly for decision-making (Stewart-Oaten 1996, McCloskey 2008). Significance testing alone, however, provides neither the magnitude of the effect nor its associated precision.

4) The focus on α=0.05 ignores or leads to low power

Power[14] is a function of the effect size, α and β values and the sample size and design. A power analysis uses the following interdependent components (Equation 1): sample size (n),

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[12] Alpha (α) is the acceptable rate of Type I errors, or incorrectly rejecting the null hypothesis.

[13] Beta (β) is the acceptable rate of Type II errors, or incorrectly accepting a false null hypothesis.

significance criterion (α and β), effect size (ES) and σ (population standard deviation) to determine power (di Stefano 2003, Nakagawa and Cuthill 2007).

Equation 1: $\boldsymbol{Power = (1 - \beta) \propto (ES \times \alpha \times \sqrt{n})/\sigma}$

Power analysis is not commonly addressed within the reported research design or in the subsequent statistical analyses (Toft and Shea 1983, Andrew and Mapstone 1987, Hayes 1987). Rosnow and Rosenthal (1989) argue that studies are too often done without consideration of the consequence of having low power (and the implications for Type I and II errors) as a result of a small sample size or a chosen significance level. This has significant implications for the potential for the persistence or recovery (and associated management activities) for these species with small populations (e.g., threatened or endangered species) or effect sizes (Fidler et al. 2006).

5) Misinterpretation of results due to low power creates false certainty

Low power may lead to a failure to reject the null hypothesis of "no effect" by investigators and a conclusion of "no impact" by managers and others not intimate with the investigation or statistical testing (including scientists) (Peterman 1990a, Stewart-Oaten 1996). Failure to reject the null hypothesis does not indicate that it is therefore true, however, but rather that there may not have been sufficient sample or effect size to detect a difference (Cohen 1990, Altman and Bland 1995). This common error, known as the "fallacy of the false negative" (Page 1978), results in misinterpretation of the statistical output, i.e., concluding "no impact" or "no effect" based on statistical non-significance, hence a sense of ***false certainty***. Even if a conclusion of "no impact" is not accepted outright, speculation or disagreement over a "non-significant" impact can hinder management action (Fairweather 1991, Peterson 1993). When a study states that there is no evidence of an effect, Altman and Bland (1995) emphasize the need to determine whether this is due to the true condition, or whether the study characteristics (e.g., sample and effect size) prevent the detection of an effect – and in either case, whether absence of evidence justifies absence of action. Toft and Shea (1983) assert that a "positive" finding (i.e., a statement of fact, in this case, one of no impact) based on a negative result (i.e., failure to reject the null hypothesis) should be subjected to the same rigour as a positive conclusion (i.e., a statement of fact, in this case, one of impact) based on a positive result (i.e., rejection of the null hypothesis).

[14] The power (of a test) is the probability of correctly rejecting the hull hypothesis and the complement of the Type II error rate β, 1-β (Lehmann and Romano 2005, Nakagawa and Cuthill 2007).

Potential (partial) solution: Power analysis with associated effect size

Despite its weaknesses, Neyman and Pearson created a paradigm that will continue to play a central role in statistics (Lehmann 1992), as evidenced by its status as a criterion for acceptance into many scholarly journals, particularly those in the natural sciences. Several authors, however, have suggested methods to improve the quality of data analysis and presentation, e.g., power analysis (Osenberg and Schmitt 1996, Stewart-Oaten 1996).

Power analysis has several benefits: it allows the investigator to choose the best experimental design and statistical analyses (in *a priori* analysis); can shed light on the confidence with which an investigator can draw conclusions regarding non-significant results (in *post hoc* analysis); and can save time and money by avoiding experimental design whose resulting power would have been too low to detect an impact regardless of its presence (Fairweather 1991). A power of 0.8 (or an 80% chance of correctly rejecting the hull hypothesis) is often recommended as desirable (Cohen 1965 in Rosnow and Rosenthal (1989)) and traditional power ratios typically follow the 'five-eighty convention' (di Stefano 2003), based on $\alpha=0.05$ and $\beta=0.20$ (power=$1-\beta=0.8$) (Field et al. 2004). Although this standard is still biased against Type II errors by a multiple of four over Type I errors (and their respective costs), it is still rarely achieved given the sample sizes often required (e.g., a correlation effect size of $r=0.30$ would require a sample size of 115 at $\alpha=0.05$; Rosnow and Rosenthal 1989). Rosnow and Rosenthal (1989) also advise computing effect size when the results are both (statistically) significant and non-significant, as this will identify the sample size needed for sufficient power, as well as shed light as to whether the effect may be (practically) significant, aside from its statistical (non)significance.

Risk management with Type I & II errors

In an environmental impact assessment context, Type II errors can have serious implications for the environmental object under study. The consistent use of low α levels (and resulting lower power) may create a bias against environmental protection and management, in which Type II errors are more common than Type I errors (Mapstone 1995). The only way to reduce both error types is to reduce the uncertainty (Stewart 2000). When this is not possible, one inevitably must make a trade-off between avoiding false alarms (e.g., categorizing a harmless species as highly invasive - a Type I error) and missed impacts (e.g. categorizing a highly invasive species as harmless - a Type II error) (Page 1978). Because scientists are typically most concerned about false alarms, the permissible Type I error rate is usually set at some conventional (low) level (e.g., 0.05). As a result, experimental parameters are optimised to avoid Type I errors (false alarms) creating variations in experimental parameters that primarily affect the miss rate, including decreased sample sizes and increased variance leading to increased rates of Type II errors (missed impacts).

Type II errors (missed impact) may be more costly than Type I errors (false alarms) in some circumstances, particularly in areas under conservation management or over large temporal scales (Toft and Shea 1983, Peterman 1990a, Fairweather 1991). This is because Type II errors may incur potentially irreversible environmental costs, as well as management costs (while Type I errors only incur management costs). Andrew and Mapstone (1987) provide commercial fishing as an example of this cost asymmetry for environmental risks. A Type I error will result in lower-than-necessary quotas (which, however, can and will likely be adjusted as more information is gathered; Underwood and Chapman 2003) while a Type II error will result in overfishing and potential collapse of the stock (which will not only have ecological impact, but will also have long-term economic impacts that are greater than with a Type I error – complete loss of the fishery and associated profits).

Choosing between the error types involves the use of judgment as a combination of technical fact and personal perception (Hammond 1996). A focus on a low rate of Type I errors reflects a judgment about the relative importance and consequences of the two types of errors (Page 1978). In a management context, it represents a social policy that protects hazards more than people (Fischhoff et al. 1982). Hence, it may be appropriate to reverse the 'burden of proof' via statistical analysis (e.g., focus on power and Type II errors) when uncertainty is present and precaution may be warranted (i.e., the cost of Type II errors is high or unknown) (Peterman 1990a, Gray and Bewers 1996, Underwood and Chapman 2003, Field et al. 2004).

Aligning the means with the ends: Precaution and Type II errors

The traditional method of environmental risk assessment may not provide an accurate estimate of and protection from risk due to the contrast between the 'means' and the 'ends'. On one hand, the research to inform environmental risk assessment (the means) generally uses experimental design and analysis that minimizes Type I errors (i.e., avoiding false alarms), while the identification and mitigation of threats to core values (the ends) often requires experimental design and analysis that minimizes Type II errors (i.e., avoiding missed impacts). Minimizing Type I errors is the equivalent of assuming 'innocent until proven guilty', and a risk assessment under this approach requires proof that harm will occur (Fairbrother and Bennett 1999) and makes incorporation of precaution difficult (Kriebel et al. 2001) if not impossible. Understanding this bias and re-organizing the 'means' of the risk assessment toward experimental design and analysis that identifies the probability of Type II errors (e.g., power analysis) is more in line with the 'ends' of a risk assessment (i.e., assuming 'guilty until proven innocent' and requiring proof that harm will not occur).

This realignment could occur through the use of precaution and its implementation via minimizing Type II errors in applied science (Buhl-Mortensen 1996). While conventional statistical methods based on a set α often fail to accurately identify impact (Johnston and Simmonds 1990, Fairbrother and Bennett 1999), the establishment of fixed, low β reverses the burden of proof in that it forces one to prove there is no or a very low chance of missing an impact, and consequently, is in greater accord with the principles of precaution (Underwood and Chapman 2003). That is, the level of β can be interpreted as inversely proportional to the level of precaution (Peterman and M'Gonigle 1992).

Establishing the use of β as a method to incorporate precaution would also quell much of the criticism that cites the use of precaution as non-scientific or too variable and ambiguous in its definition and application (Peterman and M'Gonigle 1992, Underwood 1997). This is significant in areas beyond scientific research, particularly the WTO SPS Agreement. The use of a standardized statistical procedure (i.e., focus on a fixed β) would allow risk assessments to be more precautionary yet still fall within the mandates of the Agreement.

Not only would the use of β be amenable to the conventions of the scientific method (Gray and Bewers 1996), the careful analysis of Type II errors and power will also improve the quality of scientific enquiry, in general (Fairbrother and Bennett 1999). Rejecting the null hypothesis (or, as is common, concluding that the null hypothesis is true; see "Misinterpretation of results due to low power" in above section) without any understanding of the power to detect a difference, given the sample design, is not a very defensible or objective method to present results (Buhl-Mortensen 1996). A description of the results and their significance, along with the associated power, would provide the reader and/or user with a better understanding of their potential importance and implications, as well as providing guidance on whether or not the results should or can be used as the basis for management decisions (Peterman and M'Gonigle 1992).

Power analysis for aquatic nonindigenous species impact studies

Power analysis is a little-used approach in ecological research (Toft and Shea 1983), despite the repeated discussion and calls for its use (Andrew and Mapstone 1987, Schmitt et al. 1996). For example, Fidler et al. (2006) found that 92% of articles from *Conservation Biology* and *Biological Conservation* in 2005 did not report statistical power and 63% of articles equated statistical non-significance with "no effect" or "no relationship". Peterman (1990b) found that 98% of articles on fisheries and aquatic science with non-significant findings did not report power and 52% of these interpreted the failure to reject the null hypothesis as an indication of its truth. In a review of 120 articles in five ecology and evolutionary biology journals, Yoccoz (1991) didn't find any articles that included power analysis, nor did Fairweather (1991) in a review of 40 environmental impact statements. Jennions and Moller (2003) reviewed 697 papers from 10 behavioural ecology journals, looking at the power to detect effects for the first and last tests reported. These tests, on average, could detect a small effect 13-16% of the time and a medium effect 40-47% of the time. They also found that none of the 533 non-significant finds had an associated power reported, as well as the presence of a positive correlation between power and significance of results.

ANS have been identified as one of the primary threats to the marine environment due to their significant numbers (Kolar and Lodge 2000, Elvira 2001, Hewitt and Campbell 2008) and impacts on core values (National Research Council 1996b, Carlton 2001, Hewitt 2003b, Hewitt and Campbell 2007, Campbell 2008). Consequence (along with likelihood) assessments are used to generate risk assessments, which facilitate prioritization of resources for effective management of ANS to reduce the risk of ANS entering, establishing, spreading, and having impacts. However, low power presents a particular problem to ANS research, as studies with small effect and sample sizes are common yet may be important to management efforts given the overall paucity of research. Peterman (1990b) calls for a review of impact assessment to determine the frequency with which power is reported and (where power is not reported and the results are 'non-significant') the power and detectable effect size, via *post hoc* power analysis (Peterman and M'Gonigle 1992).

In this chapter, I use ANS impact research as a case study to review power and effect size for both significant and non-significant results of impact studies via *post hoc* power analysis. I specifically evaluate algal and crustacean effects on the abundance of other species in the community. Understanding the values of, and relationship between, power and effect size in a representative subset of ANS impact studies will benefit risk research and management outcomes. Understanding if and when low power is present will allow investigators to adjust experimental design or analysis, and managers to make decisions and create/modify policy with a greater understanding of the potential relative rates and costs of Type I and II errors. I conclude with recommendations to guide researchers and managers in making these adjustments.

Methods

Given the large number and diversity of ANS and their impacts, I chose a subset of species and impacts to include in the *post hoc* power analysis. The species were restricted to the groups algae and crustacea associated with vessel biofouling and represent a quarantine concern for Australia (i.e., absent from or under control in Australia and have the potential for introduction, establishment and spread, and for unacceptable consequences; Hayes and Sliwa 2003, Hayes et al. 2004). Algae and crustacea were chosen based on the significant number of studies focusing on these taxa (e.g., Thomsen et al. 2009, Ruiz et al. 2011). Vessel biofouling was chosen given its status as the vector with the greatest potential to transfer ANS (via the variety of "niche" biofouling areas, e.g., sea-chests, bow and stern thrusters, propellers, propeller shafts and rudder areas; Lewis 2002, Coutts and Dodgshun 2007). Thus, it will provide logistically-appropriate limits for the review while still covering a large suite of species (42.6% of species with an invasion history have life history characteristics associated with vessel biofouling; Hewitt and Campbell 2008). A list of the algae (6 species) and crustacea (45 species) matching these characteristics are provided in Appendix C: Tables 1 and 2.

For each species, a search of published literature was completed using CrossSearch (a search engine that searches across databases), ASFA (Aquatic Sciences and Fisheries Abstracts) and Google Scholar, with the keywords "Genus species" (of each species) AND "invas*" OR "native" OR "abundance" OR "density", to find experiments that focused on the effects of the respective species (alone), focused on abundance effects (measured via percent cover or density) and used manipulative techniques in the laboratory or field. Abundance was chosen based on the review by Schaffelke and Hewitt (2007) of 60 case studies on introduced algae to detect patterns of impacts that found documented impacts for 17 algae species. The impact types described in this study (and associated number of case studies) were: effects on competition (43) evidenced by high abundance of nonindigenous species, space monopolization and reduced abundance of native macroalgae; biodiversity (9); fish and invertebrate fauna (13) generally via number or abundance; other biota (via toxicity; 6); and habitat change (7). As abundance was the most frequently found impact type, it was selected to maximize the potential sample size (given the general scarcity of ANS impact data). Manipulative experiments were selected because they are given the most weight by risk assessors and decision makers, and they also tend to have issues with low power, given the common presence of few replicates and high variability (Thomsen et al. 2009). Lists of literature matching these characteristics for algae (19 articles) and crustacea (13 articles) are provided in Appendix C: Tables 3 and 4.

Each article was reviewed (and grouped accordingly) based on findings that were (1) significant, (2) non-significant[15] but could be used to calculate power and (3) non-significant and could not be used

[15] P*ost hoc* power analysis is only relevant for non-significant results, i.e., significant results cannot be questioned when there is low power, as the significance has essentially shown the test powerful enough to detect a difference (Fairweather 1991).

to calculate power. For the power analyses of findings in the second category, I used G*Power3 (Faul et al. 2007), a program used in other aquatic ecology studies for calculating power (e.g., Shanks and Shearman 2009, Large et al. 2011). The G*Power3 program uses (and can calculate based on means and/or variance) Cohen's effect size d (for t-test on means) and effect size f (for F-test ANOVAs). For other parameters required for each type of analysis, see: http://www.psycho.uni-duesseldorf.de/aap/projects/gpower/ gpower-tutorial.pdf. Several assumptions were made in the analysis. For repeated measures ANOVAs, correlation between measures was assumed 0.5 when not provided (as suggested by G*Power3). Also, given statistical analyses completed within reviewed studies required assumption of homogenous variance to be met (Student's t-test and ANOVA), I assumed this was true to allow *post hoc* analysis (which also requires homogenous variances). When variances were clearly unequal, power was not analysed.

Post hoc (also known as retrospective or observed) power analysis can be controversial due to the need to make several assumptions in the *post-hoc* process (e.g., assuming the sample and actual population effect size are identical; Zumbo and Hubley 1998) and has been met with resistance when the *post-hoc* outcomes are used to draw incorrect conclusions (Thomas 1997, Morrison 2007). As the goal of this review is to provide a survey of power for a real subset of ANS impact studies (sensu Jennions and Moller 2003), it is appropriate and acceptable to determine power based on the demonstrated (sample) effect sizes (Morrison 2007). These values (and the associated estimations of power) provide examples of expected Type II error rates for a range of parameter sets (i.e., sample size, effect size, variance and statistical methods reviewed herein) that are critical and readily available to scientists, managers and other stakeholders when making management and policy decisions. Thus, assuming effect sizes are the same is appropriate for this review. In addition, the methods for calculating power (i.e., based on effect size) follow recommendations by critics Zumbo and Hubley (1998), lending further support to the review methodology.

Results

There were a total of 14 algal studies (Appendix C: Table 3) with a total of 35 different significant analyses (Appendix C: Table 5). Within these, 74% showed a negative correlation and 26% showed a positive correlation between treatment variable and nonindigenous species abundance (Figure 5.1). There were a total of 11 crustacean studies (Appendix C: Table 4) with a total of 24 different significant analyses (Appendix C: Table 6). Within these, 71% showed a negative correlation and 29% showed a positive correlation between treatment variable and nonindigenous species abundance (Figure 5.1).

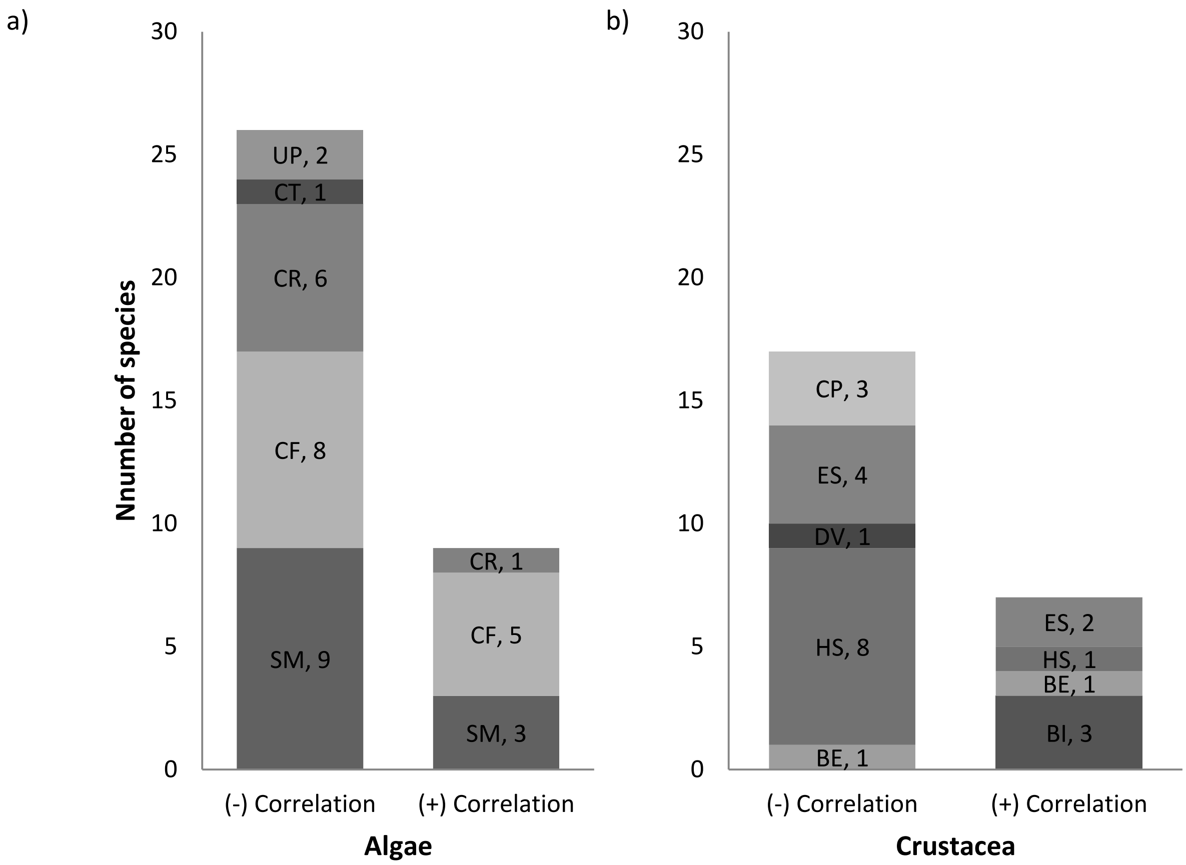

Figure 5.1. Correlation between abundance of other species and abundance of nonindigenous: a) algae; and b) crustacea species, (species name, number of analyses). For algae, SM=*Sargassum muticum*, CF=*Codium fragile*, CR=*Caulerpa racemosa*, CT=*Caulerpa taxifolia* and UP=*Undaria pinnatifida*. CT and UP both 0 for (+) correlation. For crustacea, BI=*Balanus improvisus*, BE=*Balanus eburneus*, HS=*Hemigrapsus sanguineus*, DV= *Dikerogammarus villosus*, ES= *Eriocheir sinensis*, CT= *Chthamalus proteus*. BI 0 for (-) correlation; DV and CP both 0 for (+) correlation.

There were a total of 8 articles from which power could be calculated for algae (Appendix C: Table 7), with 15 different analyses (Figure 5.2). There were a total of 3 articles from which power could be calculated for crustacea (Appendix C: Table 8), with 16 different analyses (Figure 5.2). In Figure 5.2, one data point for algae is removed (3.62, 1) as this allows greater detail via small axes for remaining analyses (with smaller values). This alteration does not change the trend of data points. There were a total of seven articles from which power could not be calculated for algae (Appendix C: Table 9) and two articles from which power could not be calculated for crustacea (Appendix C: Table 10).

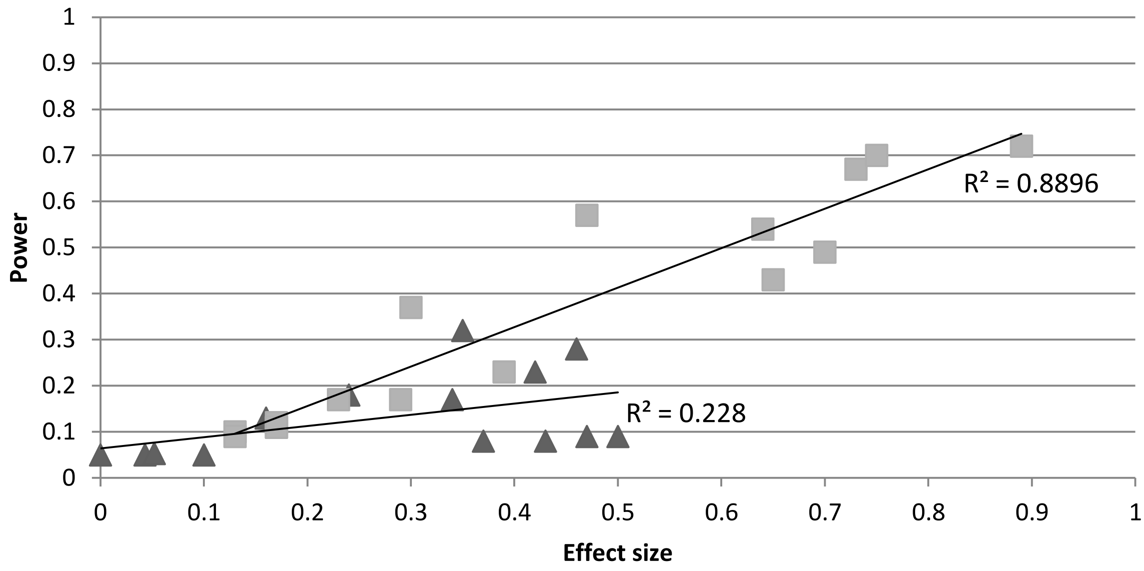

Figure 5.2. Relationship between effect size and power for the studies with non-significant findings that allowed power analysis. ▲ =algae; ■=crustaceans.

Discussion and Recommendations

The results in this chapter indicate that, using the recommended minimum criterion of 0.8 power (Cohen 1977), an extremely high proportion of the analyses (97%) had insufficient power to detect an impact. Calculating the β error rates from these power values demonstrates that, with Type II error rates of 0.28 to 0.95, these studies unconsciously accepted Type II error rates 5.6 to 19 times more than Type I error rates. Only one study, Britton-Simmons (2004), calculated power (via G*Power) and found it ranged from 0.05 to 0.4. He attributed this to the small effect size (7-12% change in percent cover) in the results but did not mention this in the discussion. These results indicate an alarming bias towards a willingness to 'miss' impacts, and therefore a bias against precaution. While the costs of these relative errors are not calculable with the available data, such an exercise would certainly provide additional insight to the consequences of high Type II error rates.

The relationship between effect size and power shows a positive trend that is greater for crustacea (r^2=0.89) than for algae (r^2=0.23). The lower value for algae may be largely due, however, to four data points from one study by Viejo (1997) for two reasons. First, the statistical analysis used t-tests, which has a different scale for the effect size categories: a value that would be considered small for an effect size f (for ANOVAs) is considered medium for an effect size d (for t-tests) (Cohen 1977). Second, this analysis had a very small sample size (4 per treatment), which may also have contributed to the lower power. Hence, the four "outlying" points may actually have a larger relative effect size than shown in Figure 5.2 (which otherwise consists of f values, save two other algal t-tests). When these points are removed r^2=0.84. Thus, both algae and crustacea show a strong positive association between effect size and power (unsurprising given the relationship between the two). More notable, however, is the low power for nearly all the studies (only one reached the 'acceptable' rate of 0.80), despite a relatively high frequency of large effect sizes. This implies that ANS may potentially have large impacts, but due to variation or sample size, are not "picked up" by statistical analyses. So, not only do significant knowledge gaps of ANS impacts lead to uncertainties that hinder management (Parker et al. 1999), but the low power suggests impact studies that find no significance may lead to a false certainty that no impact occurs, which may also prevent appropriate management action.

The analysis of the significant analyses support the meta-analysis by Thomsen et al. (2009) that determined the range of effect sizes (Hedges effect size, d) for 18 field-based manipulative experiments on 6 different algal species. They found significant negative effects on macrophyte abundance ($ES_{cumulative}$=-0.30; measured via cover and biomass of native taxa) (Thomsen et al. 2009). An interesting distinction for future field studies could occur between the status of the non-ANS treatment species (i.e., indigenous or not).

Several possible recommendations can be made based on these outcomes in order to avoid the false certainty caused by low power. Most obviously, *a priori* and *post hoc* power analyses are essential, particularly given the evidence for impact provided by the significant analyses. Second, non-significant findings without a reported power analysis should be accepted cautiously, particularly if they present a low effect size (20-50%), a gradual decline, or small sample size, as they may 'miss' an impact (Fairweather 1991). Third, non-significant results from tests with demonstrated low power should not be interpreted as demonstrating no impact, rather, that the data concerning an impact are inconclusive (Hayes 1987). Several authors (e.g., Hayes 1987, Peterman 1990b, Fairweather 1991, Underwood 1997, Strayer 1999) offer other potential solutions, including the use of an α level greater than 0.05 (such as α=0.10 or 0.25) in order to achieve greater power, particularly for small effect sizes. While useful, these considerations do not provide a clear approach for dealing with the demonstrated duality of low power and potential large effect size, particularly in situations warranting precaution or meant to serve as the basis for policy decisions. Methods for more systematically rethinking impact assessment methods are presented below.

Consideration of natural resource management in impact assessment

As impact studies generally attempt to determine if the hazard has an effect on the components and processes of an ecosystem, understanding the magnitude and direction, as well as the power to detect, any impacts should also be mandatory elements of the analysis (Mapstone 1995). Recognizing this, several authors (e.g., Bernstein and Zalinski 1983, Rotenberry and Wiens 1985, Mapstone 1995, di Stefano 2003) suggest an alternative to the standard significance testing methods based on a fixed α and Type I error rate. They suggest: (1) focusing on the effect size (or, for impact assessment, the "maximum acceptable impact"); (2) deciding on a multiplier to determine α and β based on the relative importance and costs of each error; and (3) choosing a sample size necessary to realize these values (given the interdependence of α, β, effect size and sample size and design).

1) A pre-determined effect size

   Choosing a critical effect size is a process that should be based on the situation-specific spatial and temporal conditions, as well as social and political considerations, for the range of core values and their relative importance, i.e., a significant impact level requiring management or regulatory action. Using research methods to provide results based on a pre-determined effect size that correlates with a threshold level of acceptable impact, or risk, would allow more effective policy decisions. In addition, investigators wouldn't have to fundamentally change their experimental approach, but merely ensure that experiments are designed to be powerful enough to detect the chosen effect size.

2) A multiplier to determine α and β based on their relative importance and costs

   In his discussion of choosing α and β significance levels, Mapstone (1995) suggests that an environmentally conservative option would be to give primacy to a low Type II error rate, with secondary concern to the Type I error rate. However, setting a constant β would only be a repetition of the "constant α" mindset and associated criticisms. Instead, establishing a relationship between the acceptable level of α and β *a priori,* via the acceptable level of each error (based on the specific consequences of each) would provide a procedural standard that could be applied to a variety of situations. Page (1978) suggested weighing the costs of Type I (C_I) and Type II (C_{II}) errors to determine how the balance of false alarms and missed impacts should be struck, an idea that was further developed by Mapstone (1995), such as using these relative costs to create a ratio ($k=C_{II}/C_I$) for the relative risk and use this value (k) as a multiplier, e.g., $\alpha=k\beta$. However, given the commonly-encountered difference in "currency" between the two types of risk (e.g., due to the economic implications common to Type I errors and the environmental, social, cultural or human health implications common to Type II errors), this may be a difficult process (Mapstone 1995). If the relative costs are not ascertainable, Mapstone (1995) suggests setting k=1. Setting $\alpha=\beta$ has also been described as a precautionary approach to impact assessment, and a potential tool to shift the burden of proof (i.e., ensuring sufficient power to detect an impact) on the agent responsible for the impact, where relevant (Peterman 1990a).

Given some global estimates of costs that represent Type I and II errors, however, it may be more appropriate to set the default ratio somewhat higher. Pimental et al. (2001) found losses due to invasive species represented 5% of the global gross national product. Although global research spending on invasive species is not available, the US spent 0.00169% of its 2004 GNP in this area (which represents a conservative example country given the relatively large US budget for invasive species) (Simberloff et al. 2005, Paper 2011). Using the former figure to represent Type II error costs, and the latter as Type I error costs, yields a ratio of k=2,946 (or, $\alpha=0.99966$ and $\beta=0.00034$). While these are obviously rough estimates not statistically viable, this simple exercise demonstrates the potential inequity between the two costs, and the k necessary to reflect this. Also, it should be noted that the cost of both error types may be spread across many species, in which case these costs should be divided accordingly (e.g., biofouling management actions and associated costs would cover tens if not hundreds of biofouling species). Additional research into the costs of Type II errors for a variety of species or vectors would be very helpful in making such calculations.

3) A sufficient sample size

Based on the specifications above (effect size, β, and α), the final (dependent) variables, sample size and design, can be selected. Although deviations from the anticipated experimental design and outcome (e.g., large variances) may cause the eventual error rates to be greater than anticipated, they will affect both error rates, as opposed to just the β values. If it is not feasible to achieve the sample size required to meet the parameters from the above section due to resource restrictions or other biological, ecological or socio-economic considerations, reductions in power or increases in α may be necessary (Cohen 1990, Spitz and Lek 1999, di Stefano 2001). This may be particularly true in ANS impact studies, as, in addition to uncertainty due to complexity and variability of the natural processes, ANS studies often have insufficient time or funding to design studies with large sample sizes (Peel 2005).

Consideration of biosecurity outcomes in impact assessment

While the recommendations above are useful in many natural resource management contexts, they can be modified to shift the driver from one of relative costs to the acceptable level of risk (or protection) and thus provide guidance for biosecurity-related research and management. Many countries use an acceptable level of risk (ALOR) to guide biosecurity decisions (e.g., import health standards or quarantine restrictions) (WTO 1995). In a risk assessment context, the risk in an ALOR is determined from a combination of likelihood (probability) and consequence (impact). The consequence (impact) component consists not only of the identified and predicted impact (Figure 5.3, quadrant A), but also the impact that wasn't identified or predicted (Figure 5.3, quadrant B). Thus, both of these impact types need to be incorporated into designing and analysing an impact experiment. The former source of impact could be incorporated via the predetermined effect size (step 1, above). For example, Australia defines its ALOR as very low (DAFF 2009), a categorical descriptor that may translate into a quantitative description of change (i.e., effect size) that can be determined by impact studies, via expert and stakeholder judgment and/or consequence tables.

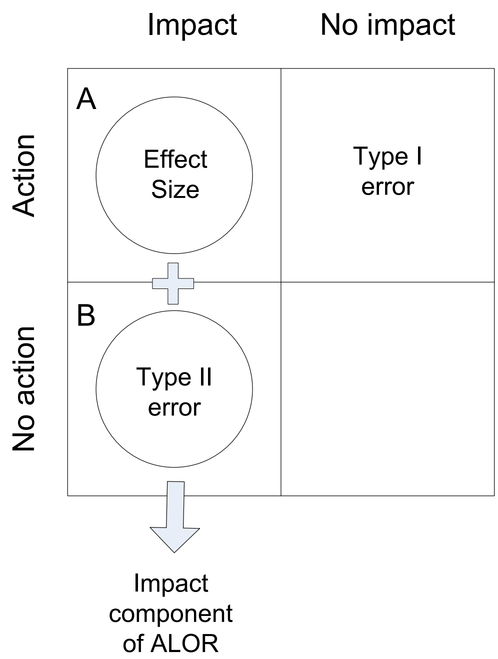

Figure 5.3. Representation of the sources of impact that contribute to the acceptable level of risk (ALOR). Quadrant A represents the identified and predicted impact, while quadrant B represents the impact that wasn't identified or predicted.

To account for the unidentified or unpredicted impact contributing to the risk in ALOR, base the acceptable level of Type II errors (β) on the ALOR and determine α based on outcomes of an *a priori* power analysis (i.e., using the established effect size, feasible sample size and observed variation from a pilot test or similar studies) (Figure 5.4).

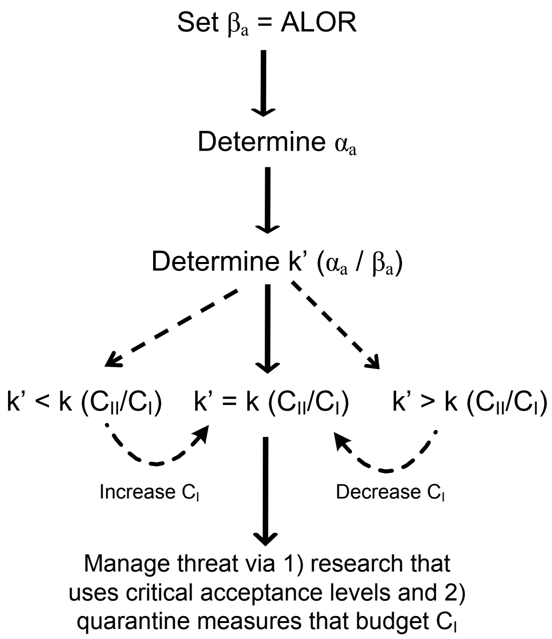

Figure 5.4. A proposed model to establish acceptance criteria based on the ALOR (acceptable level of risk) and quarantine measures spending based on the ratio between acceptance criteria. Solid lines indicate the key decision components and dashed lines indicate iterative steps that may be necessary to refine the quarantine measure spending (adapted from Mapstone 1995). β_a = acceptable Type II error rate (β); α_a = acceptable Type I error rate (α); k' = ratio between acceptance criteria; and k=ratio between costs of Type II (C_{II}) and I (C_I) errors.

From this model, compare the ratio of these levels (k'=α/β) with the ratio of the costs of each error type (k=C_{II}/C_I). Given the argument that the respective error rates should still reflect their respective costs still holds (Mapstone 1995), k should equal k'. If this does not occur, k can be modified (k' reflects the Type II error rate, β, and cannot change without adjusting the ALOR) via increasing or decreasing the cost of a Type I error (C_I). As this C_I is the cost of taking preventative biosecurity measures followed by no impact occurring, adjusting C_I is equivalent to adjusting the amount spent on these measures. Thus, this value is easily[16] adjusted to harmonize k' and k while also providing a recommended budget for preventing the incursion of an ANS.

[16] 'Easily' is in this context mathematical. However, budgetary constraints may prevent an increase in spending on preventative measures. In this case, the only way to adjust k is to decrease costs of Type II errors, which translates into eliminating the ANS threat (e.g., banning entry of ships from certain regions).

This process has several advantages for biosecurity risk assessment and management. The *a priori* focus on the effect size (via ALOR) requires prioritization of impacts on core values, and discussion and agreement on the associated impact threshold(s) that trigger action, which not only enables improved experimental design (and allows more efficient use of scare financial and other resources; Andrew and Mapstone 1987), but also facilitates discussion on a topic that is generally done *post hoc* or not at all (Stewart-Oaten 1996, Underwood 1997, di Stefano 2003). The use of the ALOR to determine β aligns the statistical-based acceptable level of risk with the policy-based acceptable level of risk. These two steps help account for both types of impact that contribute to the risk in ALOR. Finally, comparing the α/β ratio with the C_{II}/C_{I} ratio ensures the acceptable levels of each criterion reflect the consequences of both types of errors, rather than just Type I.

In addition to improving biosecurity measures by processing available and designing new research to account for the (in)ability to often fully predict impact, this process also strengthens the relationship and potential collaboration between science and management sectors. Currently, the quality and quantity of feedback is limited within what is acknowledged to be a weak relationship between management and science, with an imperfect flow of information from science to management (Peterson 1993, Schmitt et al. 1996). Research outcomes are not always presented in a manner amenable for use by management, and as a result their uptake by natural resource managers may not occur (Gibbons et al. 2008). For example, as *p*-values are a function of factors such as sample size and design, *p*-values for similar studies may appear to be contradictory if these factors are not similar, making comparison difficult or impossible (Yoccoz 1991). Specifying the parameters of the analysis will also allow for improved comparison between studies and more efficient use of public funds. Peterman (1990b) recommends that managers familiarize themselves with the issues associated with effect size, power and Type II errors, and then require scientists to report β as well as *p*-values, take care when making decisions of 'no impact', provide sufficient funds and time to achieve high-powered tests and manage conservatively or with precaution when forced to make decisions in situations of 'non-significance' and low power. As managers improve the communication of their needs to investigators in these and other areas, the investigators will be better able to produce data that is both strong statistically and increasingly useful to management (Figure 5.5).

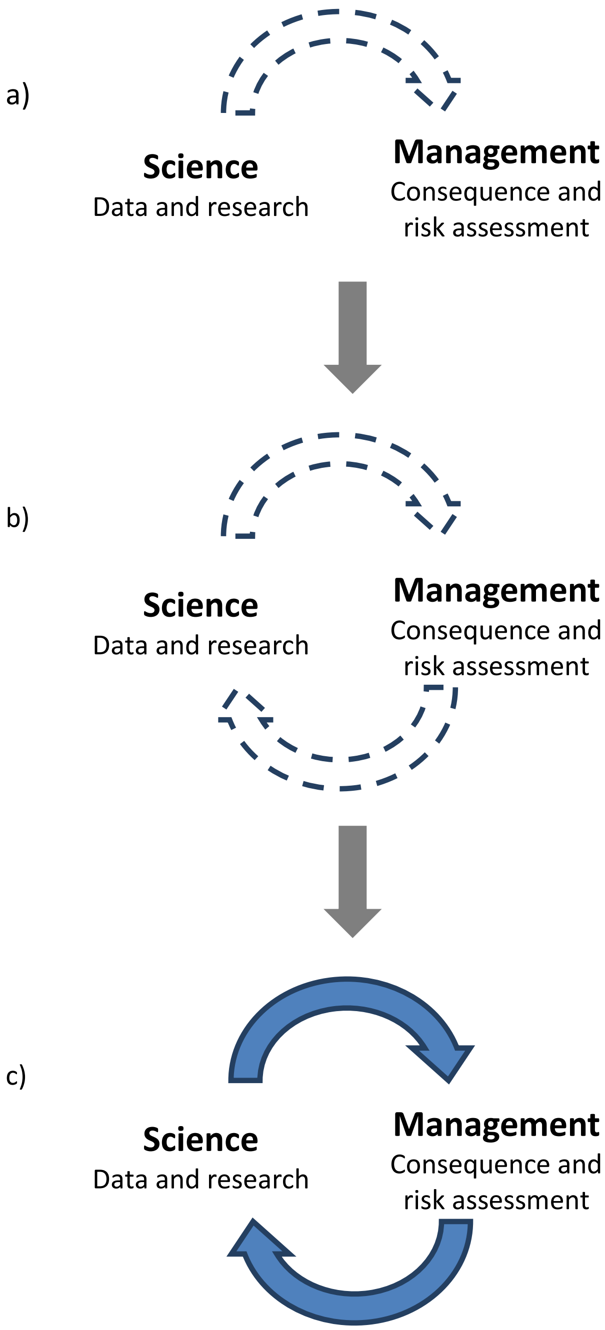

Figure 5.5. Knowledge transfer models: a)the current 'weak status' of information transfer of scientific information to inform management decisions; b)current model is improved by increased communication of management needs; and c) yields stronger information transfer between both areas.

In conclusion, these findings of low power and large effect size in non-significant statistical analyses of impact suggest that false certainty may cause scientists and managers to substantially underestimate the impacts of ANS. This false certainty, resulting from a focus on avoiding Type I errors, severely weakens science-based efforts to protect the core values from ANS impacts. The outcomes also underscore the importance of not associating absence of evidence with absence of impact, and thus absence of action. Instead, when gathering data to inform policy decisions, I suggest an alternative that translates policy needs into statistical parameters to prevent false certainty from inadvertently undermining efforts to provide sufficient protection from nonindigenous species impacts.

CHAPTER 6. GENERAL DISCUSSION

Synthesis and implications for management

This research found two sources of information used in expert decision-making under uncertainty (personal opinion and available research) that influenced risk outcomes in a non-precautionary, or "hindsight" manner (nonindigenous species assumed innocent until proven guilty). This occurred despite the mention or endorsement of precaution by the majority of biosecurity risk assessments reviewed and views expressed by the experts surveyed. A non-precautionary outcome from expert-decision making occurred due to the assignation of low consequence when uncertain, potentially via a heuristic based on the scientific norm of assuming no impact without evidence (Figure 4.6). A non-precautionary outcome from the use of available research occurred due to the traditional statistical methods applied in the impact research. That is, the traditional focus on avoiding Type I errors led to a 'false certainty' of no impact when in reality, the low power led to potentially high rates of missing an impact.

The management implications of these findings are considerable. If uncertainty within ANS risk assessment is to be managed in a precautionary manner, as supported by experts and suggested by the Convention on Biological Diversity and others (Campbell et al. 2009), the response to scarce information and the treatment of available information requires a shift. To mitigate the non-precautionary tendencies in expert opinion 'pre-assessment', a modified Delphic process with a variety of available stakeholders may avoid the potential biases (identified in Chapters 3 and 4) and mitigate the effects of uncertainty on consequence estimates, as evidenced by the increase in consequence estimate after this process (Chapter 3, Figure 3.1). Specifically, to ensure a comprehensive assessment, the Delphic process should involve experts in a variety of core value areas, as the experts felt they were working at the edge of their expertise when asked to describe social and cultural impacts. Experts also supported using alternative information sources including empirical evidence from other regions or from similar species, as well as non-empirical evidence (Chapter 3). 'Post-assessment', managers can use the expert consequence estimates to make enlightened decisions, given awareness of the potential effects of the hindsight heuristic (assumptions of no impact without evidence). That is, group species into management quadrants by consequence and uncertainty (Chapter 4, Figure 4.8), so that those with high uncertainty and low consequence are left in the assessment (following expert indication that ANS have impact based on their nonindigenous status; Chapter 3) and treated with precaution. Risk assessors can also account for the effects of uncertainty and transparently apply precaution via expression of a range of risk outcomes using standard errors or a full span of the consequence estimates (Chapter 4, Figure 4.9). To mitigate non-precautionary outcomes resulting from the conventional uptake of available research, research to inform risk assessment can shift statistical focus on the conventional α=0.05 to a focus on β, using the relative costs of each error type to then determine biosecurity spending (Chapter 5, Figure 5.4). That most experts (75-83%) indicated avoiding Type II errors as more important than Type I errors supports this shift (Chapter 3), as does other literature (e.g., Buhl-Mortensen 1996, Hewitt et al. 2006). The public source of most biosecurity-related funding for ANS research reinforces the importance of obtaining results useable for public benefit (i.e., risk assessment).

Given the outcomes of Chapter 2 to 5, I've developed a model for estimating species consequence given a range of information quality and quantity (Figure 6.1). Risk assessments do not occur in isolation, but are often completed as part of or due to political- and economic-related activity. As such, I use the concept of acceptable level of risk as a basis for the model to reflect policy obligations and also allow adaptability across agencies or countries. The increase in ANS numbers and impact types, budget limitations and global connectivity underscores the imperative for a risk assessment able to adapt to a variety of information sources, funding availability and policy backgrounds.

A response to uncertain risks, both new and old

While tempting to surrender hope of ANS prevention in the face of the ever-growing ease of vessel movement and the subsequent large and ever-growing threat and potential impacts of ANS, regional trade agreements may provide an opportunity for the regional coordination often been called for (Burgiel et al. 2006). For those individuals and organizations, a successful attempt at this coordination requires a clearly defined risk assessment framework (particularly for consequence assessment) that can be applied broadly.

Maguire (2004) suggests the use of a decision analysis framework analysis to help make decisions on invasive species management in situations of (1) uncertain outcomes of possible management actions; (2) many and potentially conflicting objectives for management; and (3) numerous stakeholders and their respective views. As such, and based on the outcomes of Chapters 2 to 5, I suggest a decision-making framework for ANS risk assessment under uncertainty (Figure 6.1; Table 6.1).

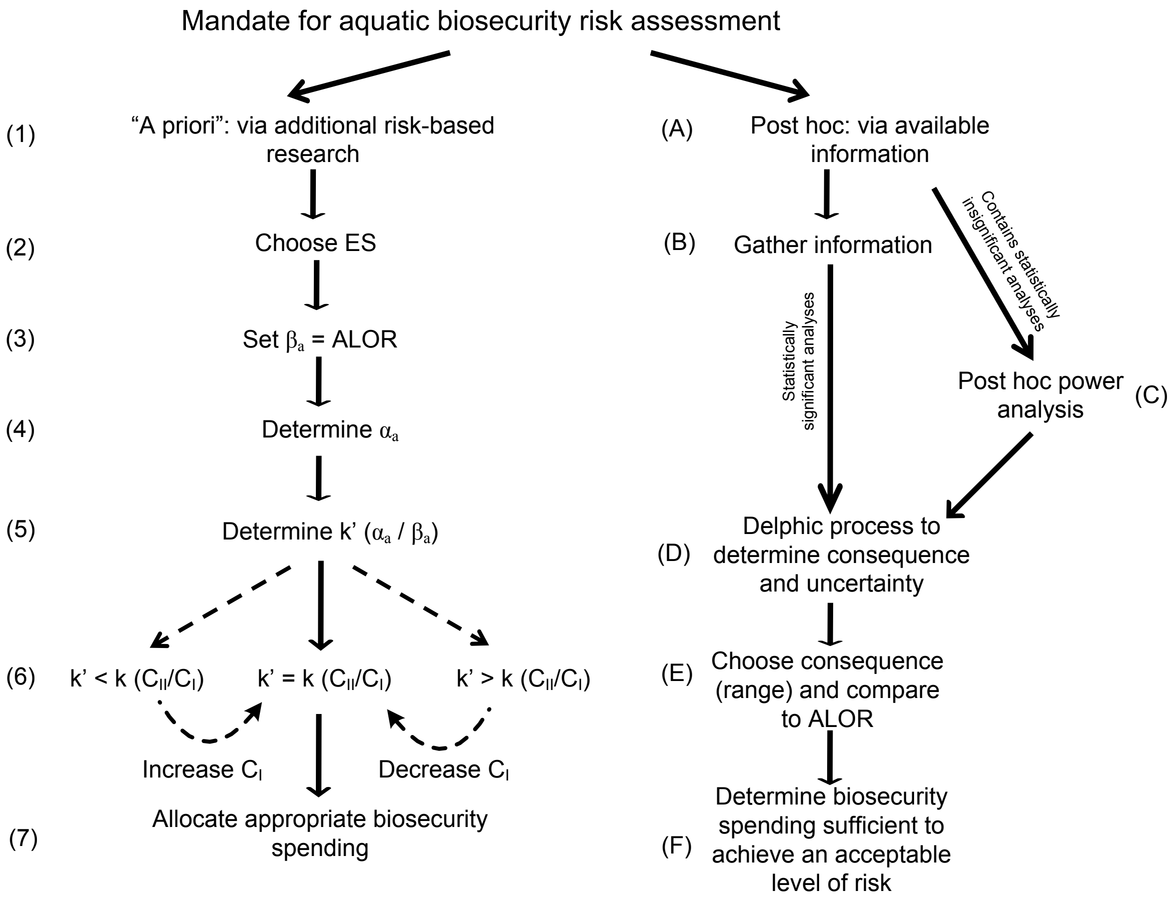

Figure 6.1. Proposed consequence assessment framework, given the mandate for a biosecurity risk assessment. ES = effect size; β_a = acceptable Type II error rate (β); α_a = acceptable Type I error rate (α); k' = ratio between acceptance criteria; and k=ratio between costs of Type II (C_{II}) and I (C_I) errors. Left side modified from Mapstone (1995). Additional description of each step provided in Table 6.1.

Table 6.1. Additional description of steps in the consequence assessment framework. The steps in *a priori* research and assessment use ALOR and costs of errors to determine appropriate statistical parameters and quarantine budget. The steps in *post hoc* assessment use available information, the Delphic process and the uncertainty consequence matrix to guide and optimize management decisions.

A priori **research and assessment**	***Post hoc*** **assessment**
(1) Appropriate when proposing pre-assessment research on a threat and designing experimental parameters to reflect and test for the acceptable level of risk and/or impact. (2) Use consequence tables to translate qualitative description of risk/impact based on ALOR into semi-quantitative descriptors of effect size. (3) Choose acceptable level of β (e.g., if ALOR is very low, β=0.05). (4) Determine α based on *a priori* power analysis, based on effect size, feasible sample size and observed variation (e.g., from a pilot test). (5) k' represents the ratio between acceptable rates of Type I and II errors. (6) Determine k based on the costs of Type I and II errors. Adjust C_I until k = k'. (7) Use C_I to determine appropriate quarantine measures. If an increase in C_I sufficient to equal k' (and thus achieve the pre-determined experimental parameters) is not possible, the proposed policy or action responsible for the risk should be designated as unacceptable, and prohibited or modified accordingly.	(A) Appropriate in situations where time or other resource limitations require the use of existing information. (B) Gather all available information, including empirical research across geographic scales and for similar species, as well as alternative information sources such as observations or grey literature. (C) Studies with insignificant results should be scrutinized with *post hoc* power analysis and account for effect size and power. (D) Use the Delphic process to choose consequence and associated uncertainty levels, based on group discussion and gathered information. Potential assumptions and biases used in assigning impact (e.g., assuming no impact if unknown) should be discussed before beginning assessment. If appropriate as per risk policy, identify appropriate assumptions or other cognitive tools for participants to use when assigning impact (e.g., when studies present varying levels of impact, use that with the greatest magnitude). This process should not only include experts, but other relevant stakeholders as well (based on the core value under consideration). (E) Based on outcomes of assessment, use the consequence and uncertainty matrix from Chapter 4 to determine the consequence level. Risk policy should provide methods for how to choose this value (e.g., use highest, lowest, average, or mode assigned consequence). (F) If a budget necessary to reduce the risk is not available, the proposed policy or action responsible for the risk should be designated as unacceptable, and prohibited or modified accordingly.

This framework does several things necessary for use in a biosecurity context. It provides a transparent process and usable outcomes that: (1) integrate scientific process and management objectives; (2) are accountable for and unimpeded by uncertainty; (3) consider the assumptions used by the experts making the assessment; (4) can be adapted according to varying strengths of precaution desired by management; (5) follows WTO SPS Agreement mandates; and (6) are feasible given time and budget constraints.

1) Integrates scientific process and management objectives

When their potential application includes biosecurity- or trade-related policy, risk assessment outcomes must consider policy and management needs. Instead of the scientific analysis and management priorities remaining compartmentalized, an assessment that considers the respective objectives of both will provide an outcome amenable to both. For example, management often has difficulty making decisions under significant uncertainty (Jenkins 1996), including uncertainty due to interpretation of a risk assessment. If policy mandates a threshold level of risk or impact, communicating and developing an understanding of these parameters with the scientific community will allow for their integration into the assessment, thus providing (relatively) increased clarity for decision-making (Harlow 2004). Specifically, the framework must take into account the acceptable level of protection (or risk). For all its powers, science does not define terms like “acceptable” or “reasonable” (Crawford-Brown et al. 2004). Several agencies argue for the separation of risk assessment and risk management, but this research suggests that an *a priori* consideration of ALOR optimizes parameters such as statistical power and therefore reduces uncertainty (Figure 6.1).

An additional benefit of this integration is a transparent and supported process for setting and allocating items within a budget. Literature often focuses on tradeoffs between management activities based on cost-benefit analyses (Horan et al. 2002, Sharov 2004, Saphores and Shogren 2005, Fernandez 2008), but rarely integrates acceptable risk and uncertainty. This decision framework applies all of these factors in producing a suggested budget for management of the vector or species under consideration.

2) Accountable for and unimpeded by uncertainty

The sources of uncertainty challenging ANS risk assessment have been well-documented (Chapters 2 and 3), as has the importance of describing this uncertainty (Crawford-Brown et al. 2004). Due to this uncertainty, decisions to manage or ignore a vector with potential to introduce ANS are subject to error. Not only are the prices of these errors likely to be asymmetrical, with greater cost resulting from mistakenly ignoring a potential ANS (Buhl-Mortensen 1996), so too, are the distribution of those costs likely to be skewed, with the environment and public bearing much more than those responsible for the introduction (Maguire 2004). As such, there is a need for a framework that considers the cost of both types of errors and can provide outcomes despite uncertainty (given that an 'unknown' designation is not useful for management decisions and often interpreted as 'no risk').

3) Considers the assumptions used by the experts making the assessment

Those responsible for risk assessment often fail to consider the assumptions on which the experts involved base their judgments (National Plant Board 1999, Harlow 2004). Defining the assumptions resulting from experts' different worldviews will improve the risk assessment process (Harlow 2004) by identifying and (if appropriate) separating, or subjugating, these subjective influences in the decision-making process (Maguire 2004). The benefits include a more transparent and repeatable process and outcome, as well as a more harmonious and efficient discussion and interaction between the experts assessing the hazard.

4) Adaption according to varying strengths of precaution desired by management

Currently, most ANS management decisions are made based on the assumption that a species will not cause harm, unless sufficient evidence indicates otherwise – that is "innocent until proven guilty". The reason for the continued application of this dogma likely lies in the inertia gathered from a long history of use, as well as fiscal realities. The long list of species causing unpredicted environmental, economic, social, cultural and human health impacts would surely belie any claim that this is an optimal solution for environmental conservation (e.g., Williamson 1996, Mack et al. 2000). Indeed, the call for a reverse assumption, that is, "guilty until proven innocent", has been made repeatedly (Campbell 2001, Simberloff 2005), and primarily for the most anthropocentric of reasons: economic gain.

Despite arguments to the contrary, precaution in preventing ANS is often cheaper (Campbell 2001). Keller et al. (2008) tested the economic costs and benefits that would have been derived from choosing different thresholds to guide management of an invasive crayfish for several inland lakes, based on the net value represented by the difference between the cost of protecting a lake against the impact of the crayfish. They estimated the net value for several management strategies, that ranged from a low threshold of risk (most lakes were protected) to a high threshold of risk (few lakes were protected). They found low management thresholds produced financial gains of $32.8 million, a significantly greater total value over the 30-year time period than for any other policy (Keller et al. 2008). In addition, they suggested the net benefits of prevention efforts were likely underestimated for several reasons, including the benefits of management efforts would be greater than indicated because of the protection provided from other invasive species (Keller et al. 2008).

Economics notwithstanding, a degree of precaution will at least slow the human-induced degradation of ecosystems around the world by introduced species. Given the extensive services provided by these same ecosystems, such an effort will not only ensure our own survival but show at least some sign of respect for the non-human flora and fauna of the world.

5) Follows WTO SPS Agreement mandates

Available scientific evidence and economic analysis aside, any biosecurity activity that may impact trade must, for now, fall in accord with trade mandates, namely those of the WTO SPS Agreement. More often than not, precaution and the 'guilty until proven innocent' approach have been seen as failing in this respect (Campbell 2001). As such, the development of the framework provided included careful consideration of each of the relevant WTO principles and obligations to ensure the validity of any resulting biosecurity measure:

- National sovereignty. The framework allows Members to consistently and clearly apply their chosen acceptable level of protection (ALOP).

- Scientific principles and evidence. This is perhaps the most-oft cited principle in disputes of SPS measures, particularly those that may intend to apply some degree of precaution. Campbell (2001) states that to be 'science-based', a phytosanitary program should reflect the serious threat of ANS. The components and rationale for the framework provided (particularly those related to precaution) are based on input from scientific and natural resource management experts and reflect this threat. Bernstein (1983) supports the view that the rationality within scientific risk estimates is found in discussion between scientists of the content and process of the assessment.

- Harmonization. The review in Chapter 2 found insufficient guidance for Members to conduct a consequence assessment, particularly in conditions of uncertainty. This framework potentially provides a process for consistent consequence assessment across countries.

- Risk assessment. This process facilitates more effective and higher rates of completed risk assessments through flexible demands of time, resources and information.

- Transparency. This process clearly identifies the steps and their associated rationale, to allow a clear understanding by other Members.

6) Feasible given time and budget constraints

As a risk assessment is often the preliminary step necessary to trigger appropriate management efforts, a framework that doesn't require extensive resources is necessary for protection of threatened core values (Burgiel et al. 2006). Yet risk assessments under WTO standards are often expensive and time consuming (Lovell and Stone 2005). For example, risk assessments by US federal agencies can cost $500,000 (US Department of Agriculture 1991). The US Animal and Plant Health Inspection Service stated it considered solid wood packing material as one of the biggest threats in May 1998, but only released the risk assessment in late 2000, with an expected ruling five years after the threat was first recognized (Campbell 2001). In light of the increasing rates of trade, introductions and their synergies with other threats such as climate change, assessing the risk of trade policies must be achievable within shorter timeframes and limited budgets in order to keep up (Hulme 2009).

A risk assessment that is flexible in terms of required expertise, information and budget is particularly important for small island and developing countries and territories that often lack these resources (Mumford 2002, Burgiel et al. 2006), particularly given the increasing amount and variety of trade by these countries (Jenkins 1996). Despite a desire for biosecurity, the ability to become highly biosecure may just not be possible (Smith 1997). As a result of, and impetus to improving this situation, island ecosystems make up some of the most impacted and threatened in the world (Donlan and Wilcox 2008). Also contributing to the risk from ANS is the difference in imports as a percentage of GDP (given this factor's influence on invasion risk; Perrings et al. 2002): the average for island countries is 43%, continental countries 27%, and the overall average (of 26 countries) 32% (Dalmazzone 2000).

However, a risk framework amenable to developing or other limited-budget countries remains wanting (Harlow 2004). This lack of consideration has resulted in part from a 'lowest common denominator' of biosecurity protection that often fails to protect vulnerable countries (Burgiel et al. 2006). In addition to the environmental, economic and social injustice this represents, introduced species are a 'weakest link' phenomenon; one 'bioinsecure' country raises the risk for all (Perrings et al. 2000). Harlow (2004) has suggested that the limited ability by developing and island countries to develop and implement biosecurity policy in the face of rapid and significant increases in trade may require the application of precaution via methods that clearly show it is not mere protectionism – an approach afforded by the framework provided.

The provided framework may be subject to a variety of criticisms. Among these is the suggested change (increase) in the threshold of statistical significance, α. To this, I suggest a consideration of the different situations in which this α is applied (Crawford-Brown et al. 2004). In one, α is held as a standard for consistency and quality in drawing conclusions in peer-reviewed journals. Resulting Type I errors are often viewed as a necessary sacrifice for upholding reliable publishing criteria and their consequences are few. In the other, α is used as a standard for determining impact, upon which conclusions of consequence are made that will have repercussions in risk management decisions; Type I errors are realized in hindsight and their consequences can be severe (e.g., a devastating invasion). Given this discrepancy, I suggest the adjustment as to how, and by what measure, significance is determined, is appropriate.

I conclude that this framework: (1) integrates scientific process and management objectives; (2) accounts for and is unimpeded by uncertainty; (3) considers the assumptions used by the experts making the assessment; (4) can be adapted according to varying strengths of precaution desired by management; (5) follows WTO SPS Agreement mandates; and (6) is feasible given time and budget constraints. As these features can be applied for both *'post hoc'* and *'a priori'* risk assessment contexts, this framework provides a widely-applicable decision framework necessary to manage the ever-changing nature of aquatic introductions.

Future direction

The work from this thesis could be expanded in several directions. Some of the most currently relevant and potentially fundable ideas include: (1) exploration of how attitudes of terrestrial biosecurity and quarantine experts and agencies to uncertainty and precaution compare with these aquatic findings; (2) more in-depth power analysis of impact studies including a broader range of impacts and taxa examined; and (3) application of the experimental (specifically, statistical) approach described in Chapter 5 to actual field impact studies to determine improvements and usefulness.

At the very least, future ANS impact researchers must reconsider how they determine and display statistical outcomes. Given the small sample and effect size common to studies of nonindigenous species impacts, the focus on Type I errors and inattention to power is inappropriate. Studies funded via public monies should be required to discuss outcomes in terms of Type II errors and, in some cases, the acceptable level of risk or threshold consequences. This will ensure those unfamiliar with or simply forgetful of the assumptions used in frequentist statistical analyses (which include individuals in science, as well as policy) will properly consider the evidence and have the ability to apply it in a precautionary manner if desired. Alongside this, biosecurity risk assessments that include experts, particularly scientific experts, to inform risk outcomes, must take either pre- or post-assessment steps to account for the assignation of lower consequence due to uncertainty. While perhaps appropriate in other management areas, the high costs and irreversibility of Type II errors in biosecurity management actions necessitate a different, more precautionary, approach.

Conclusion

The outcomes of this thesis contribute to biosecurity risk assessment and management in several ways, primarily through closing the identified gap in existing frameworks around the treatment of uncertainty and precaution. The outcomes demonstrate that adding uncertainty estimates after making the consequence estimate may lead to under-management of ANS. The influence of uncertainty must be addressed up front, during the decision-making process, particularly if precaution is desired. A modified Delphic process that includes experts and stakeholders with a variety of backgrounds can assist in this process. In demonstrating that uncertainty does not always lead to higher consequence estimates, this thesis also adds to existing decision-making and risk perception research. Finally, the outcomes highlight that even when we have ‘sufficient’ evidence, its use at face value may not provide the full or correct picture due to low power or other considerations. In offering potential solutions and guiding frameworks, this thesis aims to provide useful means to a very important end: understanding both existing and novel ANS threats, in order to maintain the biotic and abiotic integrity of all shared values.

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Appendix A contains a published article and has been removed due to copyright or proprietary reasons

Alisha Dahlstrom , ChadL.Hewitt , MarnieL.Campbell A review of international, regional and national biosecurity risk assessment frameworks. Marine Policy 35 : pp. 208-217.

doi:10.1016/j.marpol.2010.10.001

Appendix B1: Table 1. Alternative information types considered by survey participants.

Primary literature: – Direct empirical evidence (laboratory) (e.g., controlled experiments with quantified impact) – Direct empirical evidence (field) (e.g. before-after-control-impact designs) – Extrapolation: experimental observations outside the region under consideration – Meta-analysis: Analysis of multiple data sets may be stronger than a single, controlled study due to the variety of information sources. A meta-analysis may also be necessary when time or other resources are insufficient to set-up and conduct a new experiment (Byrd and Cothern 2000)
Impacts that are published but do not cite experimental analysis
Expert judgment and opinion
Grey literature (e.g. websites, policy documents, databases, technical reports)
“Anecdotal” information (e.g. news stories, community newsletters)
Incomplete and/or unfinished scientific studies
Personal communications with scientists
Supported/verified observations (e.g. multiple reports from individuals knowledgeable about ANS management such as restoration planners, fisheries specialist, biosecurity managers, or park directors)
Unsupported/unverified observations (e.g. a single report from an individual knowledgeable about ANS management such as restoration planners, fisheries specialist, biosecurity managers, or park directors)
Lay knowledge (e.g. observational data from the experienced public such as port managers, long-term residents of a site, fishermen)

Appendix B1: Table 2. Precaution has many definitions, often grouped into three versions: weak, moderate and strong (Cameron 2006), as well as an unclear interpretation by the WTO.

Weak	The weak version is the least restrictive and *allows* preventive measures to be taken in the face of uncertainty, but does not *require* them (e.g., United Nations General Assembly 1992). To satisfy the threshold of harm, there must be some evidence relating to both the likelihood of occurrence and the severity of consequences. Factors other than scientific uncertainty, including economic considerations, may provide legitimate grounds for postponing action. However, not all forms require consideration of the economic costs of precautionary measures. Under weak formulations, the requirement to justify the need for action (the burden of proof) generally falls on those advocating precautionary action. No mention is made of assignment of liability for environmental harm.
Moderate	In moderate versions of the principle, the presence of an uncertain threat is a *positive basis* for action, once it has been established that a sufficiently serious threat exists. For example, the United Kingdom Biodiversity Action Plan states: “In line with the precautionary principle, where interactions are complex and where the available evidence suggests that there is a significant chance of damage to our biodiversity heritage occurring, conservation measures are appropriate, even in the absence of conclusive scientific evidence that damage will occur" (Gummer et al. 1994). Usually, there is no requirement for proposed precautionary measures to be assessed against other factors such as economic or social costs. The trigger for action may be less rigorously defined, e.g., as “potential damage”, rather than as “serious or irreversible” damage as in the weak version. Liability is not mentioned and the burden of proof generally remains with those advocating precautionary action.
Strong	Strong versions of the principle differ from the weak and moderate versions in requiring action and reversing the burden of proof. Strong versions *justify or require* precautionary measures in the face of significant harm and some also establish liability for environmental harm on the side proposing the activity, which is effectively a strong form of “polluter pays”. For example, the Earth Charter states: “When knowledge is limited apply a precautionary approach …. Place the burden of proof on those who argue that a proposed activity will not cause significant harm, and make the responsible parties liable for environmental harm” (Cousteau et al. 2000). Reversal of proof requires those proposing an activity to prove that the product, process or technology is sufficiently “safe” before approval is granted.
WTO	While the WTO’s inclusion of precaution is still unclear (see Chapter 2), the SPS Agreement contains a clause that has been cited as a potential form of precaution; Article 5.7 states, “In cases where relevant scientific evidence is insufficient, a Member may provisionally adopt sanitary or phytosanitary measures on the basis of available pertinent information, including that from the relevant international organizations as well as from sanitary or phytosanitary measures applied by other Members” (WTO 1995). So while several arguments citing this as a method by which to incorporate precaution into SPS Standards, the WTO dispute settlement bodies have returned mixed verdicts as to the validity of this assertion.

Appendix B2. Internet survey as provided to participants.

Title: Development of an Aquatic Nonindigenous Species (ANS) Impact Assessment Framework

Introduction and Background

This survey is comprised of 32 multiple-choice questions, 3 open-ended questions, an optional comments section, and an evaluation section for 13 nonindigenous aquatic species. This survey should take 40 minute to complete. For each question, use the knowledge you already have about that item. If your knowledge is limited or you simply don't know, use your best judgment to answer the question. All responses will be kept confidential and results reported only in statistical form. Please remember that there are no right or wrong answers; I am simply interested in your opinions. Additional

Risk assessment combines the probability (likelihood) and impacts (consequences) of a threat (such as introduction, establishment, and/or spread of nonindigenous species. Risk analysis is the complete process of hazard identification, risk assessment, risk communication, risk management, and risk policy.

This survey includes questions on risk assessment, as well as questions on the use of precaution. In this survey, the term "precaution" should be taken as equivalent to the terms "precautionary approach" and/or "precautionary principle".

This survey is designed to provide responses that will improve the process and framework of impact assessment. Please answer the questions to the best of your ability. The information obtained from this survey is strictly confidential.

Questions

1. What is your participant number? (this is contained in introductory email)
2. In which country do you work?
    a. United States
    b. Canada
    c. Australia
    d. New Zealand
3. Please indicate your gender:
    a. Female
    b. Male
    c. Prefer not to answer
4. Please indicate your age:
    a. 18-25
    b. 26-35
    c. 36-45
    d. 46-55
    e. 56-65
    f. 65+
    g. Prefer not to answer
5. What is your highest level of education?

a. High school (secondary)
b. Undergraduate
c. Postgraduate by coursework
d. Masters by research
e. Doctorate

6. In what area of expertise was your highest level of education?
    a. Aquaculture
    b. Aquatic/Marine Biology
    c. Biology
    d. Ecology
    e. Economics
    f. Environmental Science
    g. Fisheries Science
    h. Natural Resources Management
    i. Oceanography
    j. Other: (please indicate area) _________________________
7. What taxonomic description best describes your background/speciality? (circle all that apply)
    a. Amphipod
    b. Ascidian/tunicate
    c. Barnacle
    d. Bryozoan
    e. Clam
    f. Copepod
    g. Crab
    h. Fish
    i. Gastropod
    j. Algae
    k. Hydroid
    l. Isopod
    m. Protozoan
    n. Worm
    o. Other: (blank)
    p. None of the above – ANS generalist
8. How long have you worked on aquatic nonindigenous species (ANS) issues?
    a. 0 – 2 years
    b. 2– 5 years
    c. 5 – 10 years
    d. 10-15 years
    e. More than 15 years: (please indicate number of years) _________
9. Does your work involve any of the following (check all that apply):
    a. Assessing likelihood of ANS entry, establishment, and/or spread
    b. Assessing impacts of ANS
    c. Communicating risk of ANS

    d. Managing risk of ANS
    e. Developing risk policy for nonindigenous species
    f. None of the above
10. How important are these next statements as a guiding principal in your life (not Important, important, extremely Important):
    a. Respecting the earth, harmony with other species
    b. Equality, equal opportunity for all
    c. A world at peace, free of war and conflict
    d. Protecting the environment, preserving nature
    e. Social justice, correcting injustice, care for the weak
11. For the following, please indicate whether you (Strongly agree, agree, disagree, strongly disagree):
    a. Humans are severely abusing the environment
    b. Nature will always possess unknowable mysteries
    c. It is important to have a sense of empathy and kinship with other forms of life
    d. The universe is a holistic, integrative system with a unifying life force
    e. Natural resources should be exploited for human use
    f. Wisdom and ethics are derived from interaction with other people
    g. The earth is like a spaceship with very limited room and resources
    h. The proper human role is to dissect, analyse and manage nature for one's own needs
    i. Humans should exercise dominion over nature in order to use it for personal and economic gain
    j. It is important for humans to be separate from and superior over other forms of life
    k. Human reason transcends the natural world and can produce insights independently of it
    l. The proper human role is to participate in the orderly designs of nature.
    m. If things continue on their present course, we will soon experience a major ecological catastrophe
    n. The so called 'ecological crisis' facing humankind has been greatly exaggerated
    o. The balance of nature is strong enough to cope with the impacts of modern industrial nations
    p. Nature is completely understandable to the rational human mind
12. In your opinion, how much of a threat are ANS, as compared to other threats to the environment:
    a. ANS are the greatest threat to the environment
    b. ANS are one of the biggest threats to the environment
    c. ANS are a moderate threat to the environment
    d. ANS are a low threat to the environment
    e. ANS are not a significant threat to the environment
13. For the following, please indicate whether you (Strongly disagree, somewhat disagree, somewhat agree, strongly agree):
    a. Technology can provide solutions to most challenges facing society
    b. Government safety assurances are not usually very accurate

c. Science is able to explain the natural world effectively
d. The potential gains outweigh potential risks to society from technological research and development
e. Government warnings of threat or risk are generally exaggerated
f. With enough time and money, research will continue to improve the quality of life
g. Risk assessments by private corporations are usually comprehensive and trustworthy
h. Environmental preservation is more important than economic growth
i. Science is able to predict future harms to the environment with high accuracy

14. Please indicate how you feel about the following statement: "There can be scientific uncertainty that is not recognized or cannot be defined (i.e. there are unknown 'unknowns')." (Strongly disagree, disagree, agree, strongly agree)
15. Please indicate which statement you most agree with:
    a. Science can provide answers to most research questions with 100% certainty
    b. Most research questions can be answered with high certainty
    c. Few research questions can be answered with high certainty
    d. No research questions can be answered with 100% certainty
16. Please indicate your opinion on the following statement: "Given enough time and money, research can reduce the uncertainties and knowledge gaps surrounding ANS and their impacts." (Strongly disagree, disagree, agree, strongly agree).
17. Please indicate which statement you most agree with:
    a. If ANS impact data contain significant amounts of uncertainty, any conclusions or decisions made based on the information are generally invalid
    b. Uncertainty in ANS impact analysis is unavoidable but can be managed to provide reliable results
    c. Uncertainty in ANS impact analysis can be avoided through proper identification and management of its sources
18. Please indicate your opinion on the following statement: "When analysing impacts of ANS, avoiding Type 2 errors (not assigning an impact when there actually is one) is more important than avoiding Type 1 errors (assigning an impact when there is actually not one)." (Strongly disagree, disagree, agree, strongly agree)
19. Please choose the statement that most closely reflects your opinion for an answer to the following question. If a statistical analysis of data for an observed pattern or impact yields an insignificant p-value, the data (please indicate which statement you most agree with):
    a. Should not be used when assessing impacts
    b. May be used with discretion if no other data is available
    c. Are valid to use when assessing impact (i.e. they may still be significant)
20. Please rank the importance of protecting each of the following values from threats by ANS, from 1 (most important) to 4 (least important):
    a. Environment/ecological values
    b. Social and cultural values
    c. Economic values
    d. Human health values

21. Do nonindigenous species have an impact due to their presence as a non-native component of the ecosystem?
    a. Yes
    b. No
22. If yes, do you agree with assigning a non- indigenous species “low impact” if there is an absence of impact literature for that non- indigenous species?
    a. Yes
    b. No
23. Before beginning this survey, were you familiar with the use of precaution (i.e. the precautionary approach and/or precautionary principle)?
    a. Yes, a clear understanding
    b. Yes, somewhat
    c. No, not really
    d. No, never heard of it
24. Which interpretation of precaution is used by your workplace?
    a. The presence of scientific certainty surrounding a threat of serious or irreversible damage **shall not prevent** the implementation of precautionary measures to prevent harm
    b. The presence of scientific certainty surrounding a threat of serious or irreversible damage is **a positive basis** for implementation of precautionary measures to prevent harm
    c. The presence of scientific certainty surrounding a threat of serious or irreversible damage **requires** implementation of precautionary measures to prevent harm
    d. The presence of scientific certainty surrounding a threat of serious or irreversible damage **allows provisional measures** to prevent harm until additional information necessary for a more objective assessment of risk is reviewed
    e. NA – my workplace does not use precaution
25. Which interpretation of precaution do you personally favour?
    a. The presence of scientific certainty surrounding a threat of serious or irreversible damage **shall not prevent** the implementation of precautionary measures to prevent harm
    b. The presence of scientific certainty surrounding a threat of serious or irreversible damage is **a positive basis** for implementation of precautionary measures to prevent harm
    c. The presence of scientific certainty surrounding a threat of serious or irreversible damage **requires** implementation of precautionary measures to prevent harm
    d. The presence of scientific certainty surrounding a threat of serious or irreversible damage **allows provisional measures** to prevent harm until additional information necessary for a more objective assessment of risk is reviewed
26. Should precaution be applied along a continuum, i.e. the more serious the potential impact, the more scientific uncertainty allowed before taking protective measures; conversely, the less serious the potential impact, the less scientific uncertainty allowed before taking protective measures?
    a. Yes

b. No

27. In general, what percentage of your work-related decisions involve precautionary measures?
    a. 0% of my decisions involve precautionary measures
    b. <10% of my decisions involve precautionary measures
    c. 10-25% of my decisions involve precautionary measures
    d. 26-50% of my decisions involve precautionary measures
    e. 51-75% of my decisions involve precautionary measures
    f. 76-100% of my decisions involve precautionary measures
28. The application of precaution is a necessary component in risk assessments to deal with the uncertainties present in the methods and information used. (Strongly disagree, disagree, agree, strongly agree)

29. How important is it to apply precaution for protection of the following categories? Please rank each category from 1 (the most important) to 4 (the least important):
    a. Environmental/ecological values
    b. Social and cultural values
    c. Economic values
    d. Human health values
30. Which of the following do you see as potential steps to integrate precaution into an ANS risk assessment (choose 3):
    a. Assume all cryptogenic species are nonindigenous; that is, if a species can't be determined to be native or not, assign non-native status
    b. Including public input regarding values and impact significance
    c. For nonindigenous species with unknown impacts, assign a "low" impact
    d. Use conservative estimates when developing and/or using model parameters
    e. In the final assessment, include even those species with low and/or unknown likelihood or low and/or unknown impact designation as possible risks
    f. If impacts for a particular nonindigenous species are unknown, use impacts from a similar species with known impacts
    g. When assessing impacts for a species using previously-documented impacts, use the impact of highest magnitude
31. Generalizing demonstrated impacts of ANS for future invasions is an uncertain process due to a variety of factors. What do you see as some of the biggest challenges to, and sources of uncertainty in predicting future impacts of nonindigenous species: (open)
32. When is it appropriate to use past impacts as predictors of future impacts for ANS?
    a. Most of the time
    b. Some of the time
    c. Rarely
    d. Never
33. Given the following combinations of evidence and uncertainty regarding the impacts attributed to an ANS, how would you rate the impact (assign no impact, assign low impact, assign moderate impact, assign high impact)
    a. **Uncertain observational/lay evidence** that the ANS would cause a serious negative impact

   b. **Strong observational/lay evidence** that the ANS would cause a serious negative impact
   c. **Uncertain experimental/scientific evidence** that the ANS would cause a serious negative impact
   d. **Strong experimental/scientific evidence** that the ANS would cause a serious negative impact
   e. **Uncertain observational/lay evidence** that an ANS **similar to that under consideration** caused a serious negative impact
   f. **Strong observational/lay evidence** that an ANS **similar to that under consideration** caused a serious negative impact
   g. **Uncertain experimental/scientific evidence** that an ANS **similar to that under consideration** caused a serious negative impact
   h. **Strong experimental/scientific evidence** that an ANS **similar to that under consideration** caused a serious negative impact
34. When assigning impacts for nonindigenous species with absent or insufficient peer-reviewed impact data, it is appropriate to also include (choose all that apply):
   a. Incomplete and/or unfinished scientific studies
   b. Impacts that are published but do not cite experimental analysis
   c. "Anecdotal" information, such as news stories
   d. Personal communication with scientist
   e. Heuristic/expert observation/experience
   f. Lay knowledge (e.g. observational data from public such as port managers, long-term residents of a site, or fishers)
   g. Supported/verified observations (e.g. data from more than one person involved in resource management such as restoration planners, fisheries specialist, or park director)
   h. Unsupported/unverified observations
   i. Grey literature (e.g. websites, policy documents, databases, reports)
35. A precautionary approach to risk assessment would examine/include all potential sources of information (including non-scientific information)? (strongly disagree, disagree, agree, strongly agree)
36. A risk assessment that used all potential sources of information (including non-scientific information) would alter the quality of the assessment by what degree? 1 (risk assessment quality would be much lower) to 10 (risk assessment quality would be much higher):
37. For risk assessment, various international organizations mandate that risk assessors: "take into account available scientific evidence; relevant processes and production methods; relevant inspection, sampling and testing methods; prevalence of specific diseases or pests; existence of pest- or disease-free areas; relevant ecological and environmental conditions; and quarantine or other treatment." Based on this description, could the use of precaution in impact assessment be seen as an acceptable tool for risk assessment?
   a. Yes
   b. No
38. What do you see as the biggest challenges to understanding and describing impacts of ANS: (open)

39. Please add any additional comments you feel may assist in understanding your views on ANS impacts, risk, precaution, or uncertainty.

Appendix B3. Survey 2 and 3[17].

This survey contains questions similar to the second part of the first survey. You have been provided with information on a set of 10 species. For this survey, please use the provided information to do the following for each species (without consulting other sources, except from those provided):

- For each of the 4 value categories (economic, environmental, social/cultural, and human health), indicate the magnitude of impact and the uncertainty surrounding your choice (i.e. how sure you are that the impact magnitude is accurate).

- If you are aware of any specific types of impacts for each of the 4 main categories, please list those

- For each species, please explain (1) why/how you chose a particular impact magnitude for each of the four categories and (2) why/how you chose any specific types of impacts and their magnitude.

Please remember that there are no right or wrong answers; I am simply interested in your opinions. All responses will be kept confidential.

What is your participant number? ___

Species: *Caulerpa scalpelliformis*

Please respond to the following questions for the species, *Caulerpa scalpelliformis*, using the information given below and the paper (cited below) that was sent as pdf in email.

Species: *Caulerpa scalpelliformis*
Common group name: Green alga
Size: Fronds 20cm tall, 3 cm wide
Diet: Photosynthetic
Reproduction: Horizontal spread through stolons; reattachment of fragments; sexual reproduction poorly understood
Mobility: Sessile
Habitat: Up to 100 m depth in bays and estuaries; exposed and protected rock, sand, sea grass beds
Vector(s): Fisheries, aquarium or ornamental release
Native: Cryptogenic
Introduced: Cryptogenic, but distribution in Indian Ocean, Australia, Brazil
Impacts: See paper (provided):

[17] Questions only provided for first species, as they were identical for each species in both surveys.

Falca C and de Szechy MTM (2005) Changes in shallow phytobenthic assemblages in southeastern Brazil, following the replacement of Sargassum vulgare (Phaeophyta) by *Caulerpa scalpelliformis* (Chlorophyta). *Botanica Marina* **48**: 208–217.

1. To the best of your knowledge, please indicate this species' overall impact on environmental values:
    a. Negligible
    b. Low
    c. Moderate
    d. High
    e. Extreme

2. For the question above, what is the level of uncertainty surrounding your choice of impact?
    a. Negligible uncertainty
    b. Low uncertainty
    c. Moderate uncertainty
    d. High uncertainty
    e. Extreme uncertainty

3. For this species, please list any specific environmental impacts that you are aware of (e.g. loss of biodiversity):

4. For this species, please explain (1) why/how you chose the environmental impact magnitude and (2) why/how you chose the specific environmental impacts and their magnitude:

5. To the best of your knowledge, please indicate this species' overall impact on economic values:
    a. Negligible
    b. Low
    c. Moderate
    d. High
    e. Extreme

6. For the question above, what is the level of uncertainty surrounding your choice of impact?
    a. Negligible uncertainty
    b. Low uncertainty
    c. Moderate uncertainty
    d. High uncertainty
    e. Extreme uncertainty

7. For this species, please explain (1) why/how you chose the economic impact magnitude and (2) why/how you chose the specific economic impacts and their magnitude.

8. For this species, please list any specific economic impacts that you are aware of (e.g. clogs power plant intake pipes):

9. To the best of your knowledge, please indicate this species' overall impact on social and/or cultural values:
    a. Negligible
    b. Low
    c. Moderate
    d. High
    e. Extreme

10. For the question above, what is the level of uncertainty surrounding your choice of impact?
    a. Negligible uncertainty
    b. Low uncertainty
    c. Moderate uncertainty
    d. High uncertainty
    e. Extreme uncertainty

11. For this species, please list any specific social and/or cultural impacts that you are aware of (e.g. reduces enjoyment of beaches):

12. For this species, please explain (1) why/how you chose the social/cultural impact magnitude and (2) why/how you chose the specific social/cultural impacts and their magnitude.

13. To the best of your knowledge, please indicate this species' overall impact on human health values:
    a. Negligible
    b. Low
    c. Moderate
    d. High
    e. Extreme

14. For the question above, what is the level of uncertainty surrounding your choice of impact?
    a. Negligible uncertainty
    b. Low uncertainty
    c. Moderate uncertainty
    d. High uncertainty
    e. Extreme uncertainty

15. For this species, please list any specific human health impacts that you are aware of (e.g. vector for human pathogen):

16. For this species, please explain (1) why/how you chose the human health impact magnitude and (2) why/how you chose the specific human health impacts and their magnitude.

17. Please indicate your opinion for the following statements:
    a. The impacts of this species can be controlled.
       Strongly agree Agree Disagree Strongly disagree
    b. The impacts of this species can be mitigated:
       Strongly agree Agree Disagree Strongly disagree

18. In general, what is your level of concern about the impacts caused by this species?

Unidentified Gastropod Species

Common group name: Gastropod
Size: 50-80mm length, 25mm width
Diet: Suspension feeder
Reproduction: Sexual, internal fertilization
Mobility: Sedentary
Habitat: mud, sand or rock substrates from low intertidal to shallow subtidal
Vector(s): Aquaculture, ballast
Native: Europe (United Kingdom to the Netherlands)
Introduced: United States (Virginia to Maine)
Impacts:
No demonstrated impact, may affect invertebrate diversity and predation rates in soft-sediment benthic communities through modification of habitat structure due to high abundance of this species' shells.

Species: *Pterois volitans*

Common group name: Fish
Size: 15-30cm average, largest recorded at 43cm
Diet: Carnivore (crustacea and fish)
Reproduction: Dioecious, external fertilization, pelagic egg mass
Mobility: Highly mobile
Habitat: Variable: inshore lagoons to offshore reefs <50 m depth
Vector(s): Aquarium trade, ballast
Native: Western and South Pacific
Introduced: Atlantic Ocean from New York south to the Bahamas, Columbia
Impacts: See attached paper:
Albins MA and Hixon MA (2008) Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. *Marine Ecology Progress Series* 367: 233–238.

Species: Unidentified Ascidian Species

Common group name: Ascidian
Size: Cone-shaped, 120mm length
Diet: Suspension feeder (phytoplankton, zooplankton)
Reproduction: Hermaphrodite, no self-fertilization; external fertilization
Mobility: Sessile
Habitat: Artificial or natural substrates in low intertidal or subtidal areas
Vector(s): Fisheries, fouling, ballast
Native: Mediterranean
Introduced: Brazil
Impacts:
This is a prominent nuisance fouler in aquaculture, specifically mussel rope culture, oyster farms and suspended scallop ropes. Documented impacts include: dramatically reduced harvests of mussels and increased processing costs due to increased handling and hoisting effort of culture ropes made heavy due to this species.

Species: *Bonamia ostreae*

Common group name: Protozoan

Size: 2-5µm
Diet: This species is an oyster pathogen
Reproduction: Asexual
Mobility: Passive
Habitat: Oysters, particulary gills, mantle, and digestive gland
Vector(s):Shellfish aquaculture, fouling, ballast
Native: Cryptogenic
Introduced: Cryptogenic, but distribution in Europe (France, the United Kingdom, the Netherlands, Spain, Denmark, Italy), Morocco, Eastern Pacific (Washington and British Columbia), Western Atlantic (Maine)
Impacts: See attached paper:
Lallias D, Arzul I, Heurtebise S, Ferrand S, Chollet B, Robert M, Beaumont AR, Boudry P, Morga B, and Lapègue S (2008) *Bonamia ostreae*-induced mortalities in one-year old European flat oysters *Ostrea edulis*: experimental infection by cohabitation challenge. *Aquatic Living Resources* 21: 423-439.

Unidentified Algae Species

Common group name: Green alga
Size: Fronds 25cm tall
Diet: Photosynthetic
Reproduction: Vegetative (asexual) reproduction
Mobility: Sessile
Habitat: Rock and sand substrates in subtidal areas of bays and estuaries
Vector(s): Fouling, aquarium release
Native: Cryptogenic
Introduced: Cryptogenic but distribution in Indian Ocean, Australia, Brazil
Impacts:
Where introduced, this alga has spread rapidly. An increase in cover of this species on deep-reef habitat has been associated with a substantial decline in the cover of sessile invertebrates, predominantly sponges, colonial ascidians and bryozoans. Within 12 months of the appearance of the alga in one area, random photoquadrats revealed that it had reached an average cover of 57 ± 10%. Over the same period the average cover of sessile invertebrates declined from 49 to 21%; no such decline was observed in reference sites. This alga has an ability to rapidly expand across continuous reef, as well as an ability to establish on non-continuous reef. It has also been found that herbivores associated with the alga's habitat are highly unlikely to graze at sufficient rates to control the spread of this species.

Species: *Maoricolpus roseus*

Common group name: Gastropod
Size: 60-70mm length
Diet: Suspension feeder
Reproduction: Sexual, internal fertilization
Mobility: Sedentary
Habitat: Fine silts, muds, sand, gravel or shell substrates from low intertidal to 200m
Vector(s): Aquaculture, ballast
Native: New Zealand
Introduced: Australia
Impacts:

No demonstrated impact, may have role in decline of native gastropod species.

Unidentified Parasite Species

Common group name: Apicomplexa (parasite protist)
Size: 1µm
Diet: This species parasitizes Ateroidea (starfish) species
Reproduction: Asexual
Mobility: Passive
Habitat: Intracellular
Vector(s): Fouling, ballast
Native: Cryptogenic
Introduced: Cryptogenic, but distribution in France, the United Kingdom, Spain, and Portugal, South Africa, Eastern Pacific (Mexico to Washington), Western Atlantic (Rhode Island to Nova Scotia)
Impacts:
Recently, this species has been found in several Asteroidea (starfish) species throughout the parasites range, and further infection of additional species may be possible. The life cycle outside the host is unknown, though it has been possible to transmit the disease experimentally in the laboratory by cohabitation or inoculation of purified parasites. In the wild, the parasite occurs throughout the year but prevalence and intensity of infection tend to increase during warmer months. There are no outward signs of infection, but this parasite becomes systemic with overwhelming numbers of parasites coinciding with the death of the starfish. Effected starfish have high mortality rates (50-70%). Several of the effected species are ecologically important in their native range, and several local populations have experience complete or functional extinction. It has not been determined what ecological effects the potential loss of the effected starfish species will have.

Species: *Ciona intestinalis*

Common group name: Ascidian
Size: Cylindrical, 100-150mm length
Diet: Suspension feeder (phytoplankton, zooplankton, organic matter)
Reproduction: Hermaphrodite, no self-fertilization; external fertilization
Mobility: Sessile
Habitat: Artificial or natural substrates in low intertidal or subtidal areas in enclosed and semi-protected bays and estuaries
Vector(s): Fisheries, fouling, ballast
Native: Cryptogenic
Introduced: Cryptogenic, but cosmopolitan distribution
Impacts: See attached paper:
Ramsay A, Davidson J, Landry T, and Arsenault G (2008) Process of invasiveness among exotic tunicates in Prince Edward Island, Canada. *Biological Invasions* 10: 1311–1316.

Unidentified Fish Species

Common group name: Fish
Size: 20-25cm average, largest recorded at 36cm
Diet: Carnivore
Reproduction: Dioecious, external fertilization, pelagic egg mass
Mobility: Highly mobile

Habitat: bays and estuaries to shallow offshore reefs
Vector(s): Aquarium release
Native: South Africa
Introduced: Indonesia, Australia
Impacts:
This fish poses a threat to fishermen, divers, wildlife inspectors, in particular, but also to any individual near the fish’s habitat, because it is venomous and people unfamiliar with the nonindigenous fish may not know this. This species can inject venom with multiple dorsal-fin, anal-fin, and pelvic-fin spines. This fish will not retreat under threat, but point their spines at the aggressor and swim forward rapidly to inflict a sting, most often to the individual’s hand. Serious wounds have also resulted from handling of newly dead specimens. The sting leads to several hours of extreme pain, depending upon the amount of venom received. Other symptoms of the sting may include swelling, redness, bleeding, nausea, numbness, joint pain, anxiety, headache, disorientation, dizziness, nausea, paralysis, and convulsions. Without immediate care, the sting can lead to complications and eventual loss of motion in the affected area. The stings of this fish have been implicated in human deaths, though whether this was the sole cause is not certain.

Appendix B4. Ethics approval letter.

COPY

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MEMORANDUM

UTAS

HUMAN RESEARCH ETHICS COMMITTEE (TASMANIA) NETWORK

FULL COMMITTEE APPLICATION APPROVAL

18 August 2009

Professor Chad Hewitt
NC Marine Conservation and Resource Sustainability
Private Bag 1370
Launceston

Ethics Reference: H10726

Development of an Aquatic Non-indigenous Species (ANS) Impact Assessment Framework.

PhD Candidate: Ms Alisha Dahlstrom

Dear Professor Hewitt

The Tasmanian Social Sciences HREC Ethics Committee approved the above project on 16 August 2009.

All committees operating under the Human Research Ethics Committee (Tasmania) Network are registered and required to comply with the *National Statement on Ethical Conduct in Human Research* (NHMRC 2007).

Therefore, the Chief Investigator's responsibility is to ensure that:

1) All researchers listed on the application comply with HREC approved application.
2) Modifications to the application do not proceed until approval is obtained in writing from the HREC.
3) The confidentiality and anonymity of all research subjects is maintained at all times, except as required by law.
4) Statement 5.5.3 of the National Statement states:

    Researchers have a significant responsibility in monitoring approved research as they are in the best position to observe any adverse events or unexpected outcomes. They should report such events or outcomes promptly to the relevant institution/s and ethical review body/ies and take prompt steps to deal with any unexpected risks.

A PARTNERSHIP PROGRAM IN CONJUNCTION WITH THE DEPARTMENT OF HEALTH AND HUMAN SERVICES

5) All participants must be provided with the current Information Sheet and Consent form as approved by the Ethics Committee.

6) The Committee is notified if any investigators are added to, or cease involvement with, the project.

7) This study has approval for 4 years contingent upon annual review. A *Progress Report* is to be provided on the anniversary date of your approval. You will be sent a courtesy reminder closer to this due date.

8) A *Final Report* and a copy of the published material, either in full or abstract, must be provided at the end of project.

Yours sincerely

Emily Keenan

for **Ethics Executive Officer**

Appendix B: Table 3. Participant demographics for US/CA scientists.

Factor		Total(%)
Nationality	U.S.	25(93%)
	Canada	2(7%)
Gender	Male	11(41%)
	Female	16(59%)
Age	18-25	1(4%)
	26-35	13(52%)
	36-45	6 (24%)
	46-55	1(4%)
	56-65	2(8%)
	65+	1(4%)
	Prefer not to answer	1(4%)
Highest level of education	Postgraduate by coursework	2(7%)
	Masters by research	12(44%)
	Doctorate	13(48%)
Area of educational expertise (this question allowed multiple selection, totals will exceed 100%)	Aquaculture	2(7%)
	Aquatic/Marine Biology	13(48%)
	Biology	3(11%)
	Ecology	6(22%)
	Economic	0(0%)
	Environmental Science	1(4%)
	Fisheries Science	1(4%)
	Natural Resources Management	1(4%)
	Oceanography	2(8%)
	Physiology	1(4%)
	Philosophy	1(4%)
	Marine Invasions Biology	1(4%)
Taxonomic specialty (this question allowed multiple selection, totals will exceed 100%)	Amphipod	5(19%)
	Ascidian/tunicate	7(26%)
	Barnacle	4(15%)
	Bryozoan	3(11%)
	Clam	3(11%)
	Copepod	4(15%)
	Crab	9(33%)
	Fish	2(7%)
	Gastropod	8(30%)
	Algae	5(19%)
	Hydroid	2(7%)
	Isopod	4(15%)
	Protozoan	0(0%)
	Worm	3(11%)
	None of the above – ANS generalist	6(22%)
	Fungi, Marine	1(4%)
	Zooplankton	1(4%)
	Echinoderms – urchins	1(4%)

Appendix B: Table 3 cont.

Factor		Total(%)
Years of ANS experience	0 – 2 years	3(11%)
	2– 5 years	5(19%)
	5 – 10 years	10(37%)
	10-15 years	5(19%)
	More than 15 years	4(15%)

Appendix B: Table 4. Participant demographics for AU scientists.

Factor		Total(%)
Nationality	Australia	16(94%)
	Australia and New Zealand	1(6%)
Gender	Male	10(59%)
	Female	7(41%)
Age	18-25	5(29%)
	26-35	8(47%)
	36-45	3 (18%)
	46-55	1(6%)
	56-65	0(0%)
	65+	0(0%)
	Prefer not to answer	0(0%)
Highest level of education	Undergraduate	8(47%)
	Postgraduate by coursework	2(12%)
	Masters by research	0(0%)
	Doctorate	7(41%)
Area of educational expertise (this question allowed multiple selection, totals will exceed 100%)	Aquaculture	0(0%)
	Aquatic/Marine Biology	13(76%)
	Biology	0(0%)
	Ecology	8(47%)
	Economic	0(0%)
	Environmental Science	2(12%)
	Fisheries Science	0(0%)
	Natural Resources Management	0(0%)
	Oceanography	0(0%)
	Microbiology	1(6%)
Taxonomic specialty (high knowledge)	Amphipod	0(0%)
	Ascidian/tunicate	4(24%)
	Barnacle	3(18%)
	Bryozoan	3(18%)
	Clam	2(12%)
	Copepod	0(0%)
	Crab	1(6%)
	Fish	1(6%)
	Gastropod	2(12%)
	Algae	2(12%)
	Hydroid	0(0%)
	Isopod	1(6%)
	Protozoan	0(0%)
	Worm	2(12%)
Years of ANS experience	0 – 2 years	8(47%)
	2– 5 years	3(18%)
	5 – 10 years	3(18%)
	10-15 years	1(6%)
	More than 15 years	0(0%)
	N/A I do not work on ANS issues	2(12%)

Appendix B: Table 5. Participant demographics for US/CA managers.

Factor		Total(%)
Nationality	U.S.	21(78%)
	Canada	5(19%)
	New Zealand	1(4%)*
	Mexico	2(7%)*
Gender	Male	15(56%)
	Female	12(44%)
Age	18-25	0(0%)
	26-35	8(30%)
	36-45	6 (22%)
	46-55	10(37%)
	56-65	3(11%)
	65+	0(0%)
	Prefer not to answer	0(0%)
Highest level of education	Undergraduate	3(11%)
	Postgraduate by coursework	3(11%)
	Masters by research	10(37%)
	Doctorate	11(41%)
Area of educational expertise (this question allowed multiple selection, totals will exceed 100%)	Aquaculture	0(0%)
	Aquatic/Marine Biology	6(22%)
	Biology	6(22%)
	Ecology	8(30%)
	Economic	0(0%)
	Environmental Science	2(7%)
	Fisheries Science	4(17%)
	Natural Resources Management	3(11%)
	Oceanography	1(4%)
	Physiology	1(4%)
	Philosophy	0(0%)
	Marine Invasions Biology	0(0%)
	Entomology	2(7%)
	Biogeography	1(4%)
	Library and Information Science	1(4%)
	Conservation Biology	1(4%)
	Environmental Education	1(4%)
	Electrical Engineering	1(4%)

Appendix B: Table 5 cont.

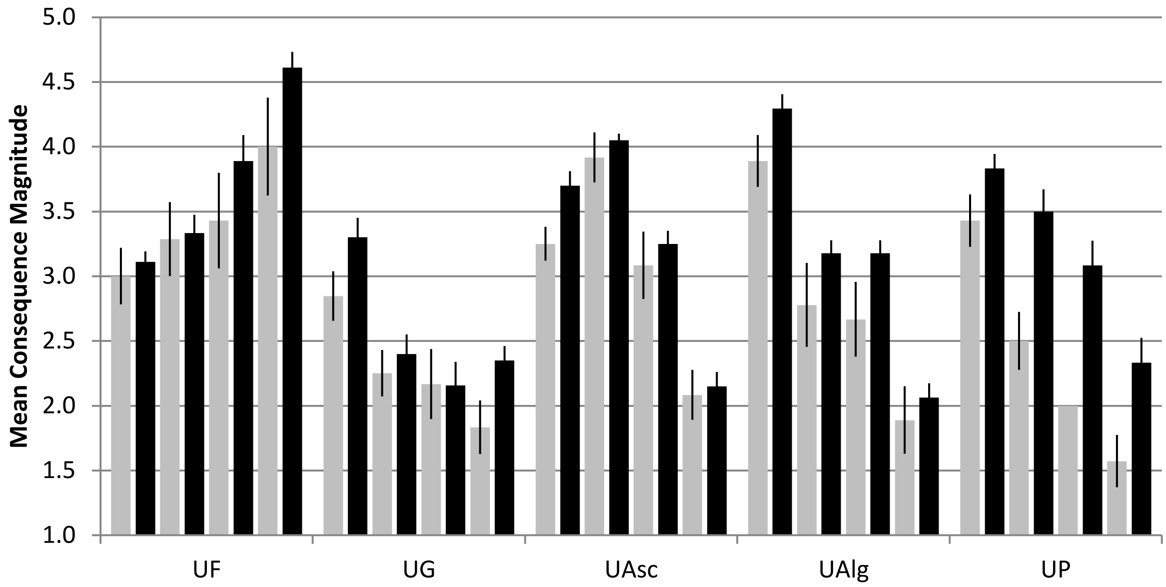
Factor		Total(%)
Taxonomic specialty (high knowledge)	Amphipod	1(4%)
	Ascidian/tunicate	2(7%)
	Barnacle	0(0%)
	Bryozoan	1(4%)
	Clam	2(7%)
	Copepod	2(7%)
	Crab	2(7%)
	Fish	9(33%)
	Gastropod	1(4%)
	Algae	5(19%)
	Hydroid	0(0%)
	Isopod	1(4%)
	Protozoan	2(7%)
	Worm	1(4%)
Years of ANS experience	0 – 2 years	7(26%)
	2– 5 years	9(33%)
	5 – 10 years	8(30%)
	10-15 years	1(4%)
	More than 15 years	2(7%; 17,19yrs)

Appendix B: Table 6. Participant demographics for AU managers.

Factor		Total(%)
Nationality	Australian	13 (100%)
Gender	Male	8(62%)
	Female	5(38%)
Age	18-25	0(0%)
	26-35	6(46%)
	36-45	3(23%)
	46-55	3(23%)
	56-65	1(8%)
	65+	0(0%)
Highest level of education	Undergraduate	2(15%)
	Postgraduate by coursework	7(54%)
	Masters by research	0(0%)
	Doctorate	4(31%)
Area of educational expertise (this question allowed multiple selection, totals will exceed 100%)	Aquaculture	2(15%)
	Aquatic/Marine Biology	1(8%)
	Ecology	3(23%)
	Environmental Science	4(30%)
	Philosophy	1(8%)
	Commerce	1(8%)
	Public Policy/Public Sector Mgmt.	1(8%)
Taxonomic specialty (high knowledge; this question allowed multiple selection, totals will exceed 100%)	Clam	1(8%)
	Copepod	1(8%)
	Crab	15(2%)
	Fish	3(23%)
	Gastropod	1(8%)
	Algae	2(15%)
	Protozoan	1(8%)
Years of ANS experience	0 – 2 years	4(31%)
	2– 5 years	4(31%)
	5 – 10 years	5(38%)
	10-15 years	0(0%)
	More than 15 years	0(0%)

Appendix B5: Figure 1. Consequence estimates for unknown species.

a)



b)

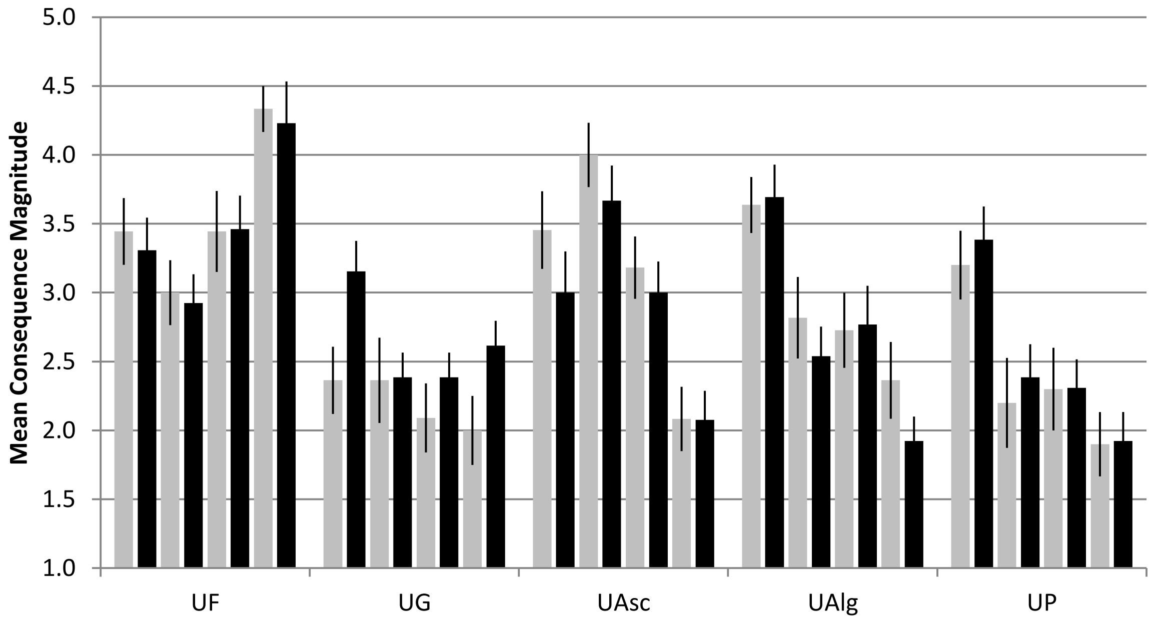
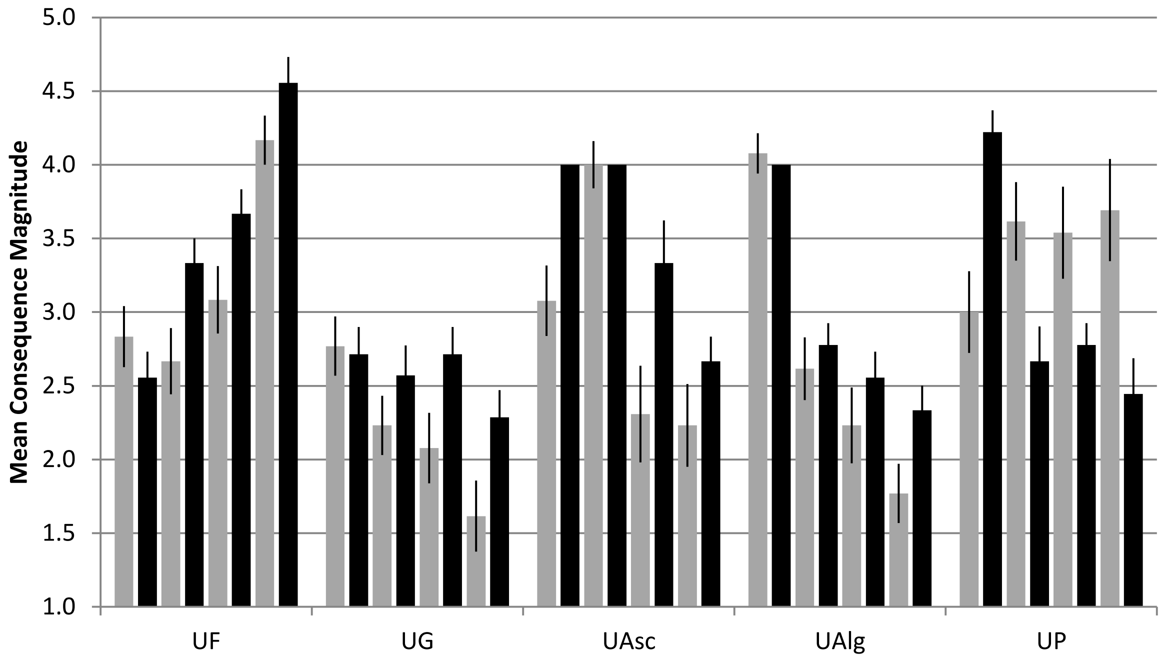


Figure 1. Average consequence for each core value, for 'unknown species' for a) US/CA scientists; b) US/managers; (overleaf) c) AU scientists; and d) AU managers. UF=Unknown Fish; UG=Unknown Gastropod; UAsc=Unidentified Ascidian; UP=Unknown Parasite; UAlg=Unknown Algae. For each species, two bars at left=environmental; two bars second from left=economic; two bars second from right=social/cultural; and two bars at right=human health; grey=second assessment; and black=third assessment.

c)



d)

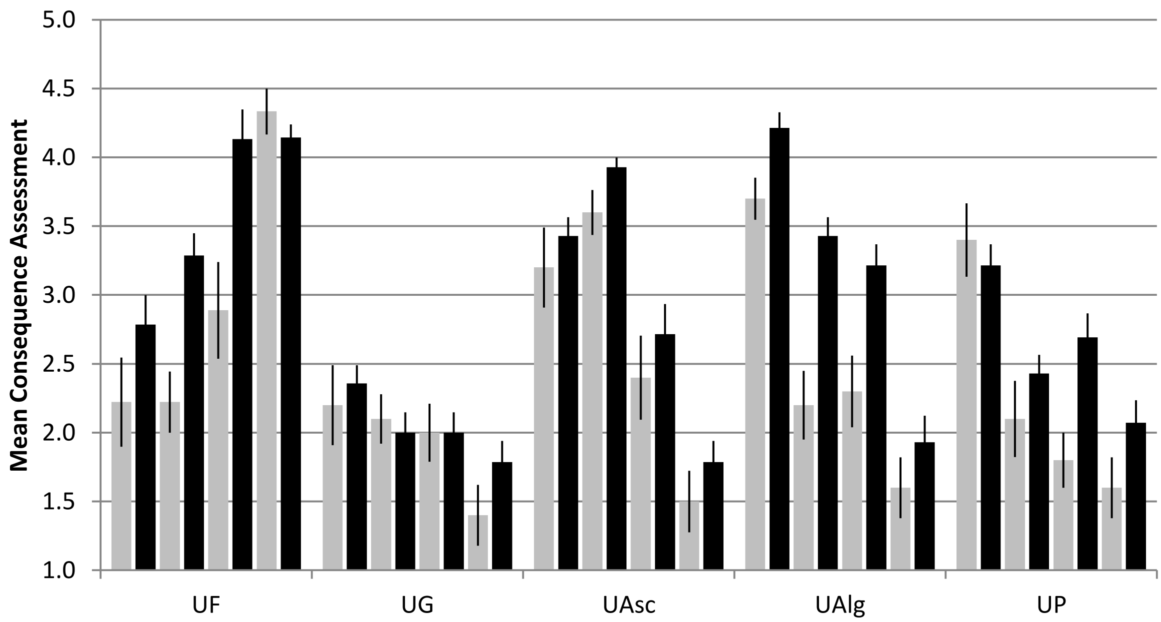
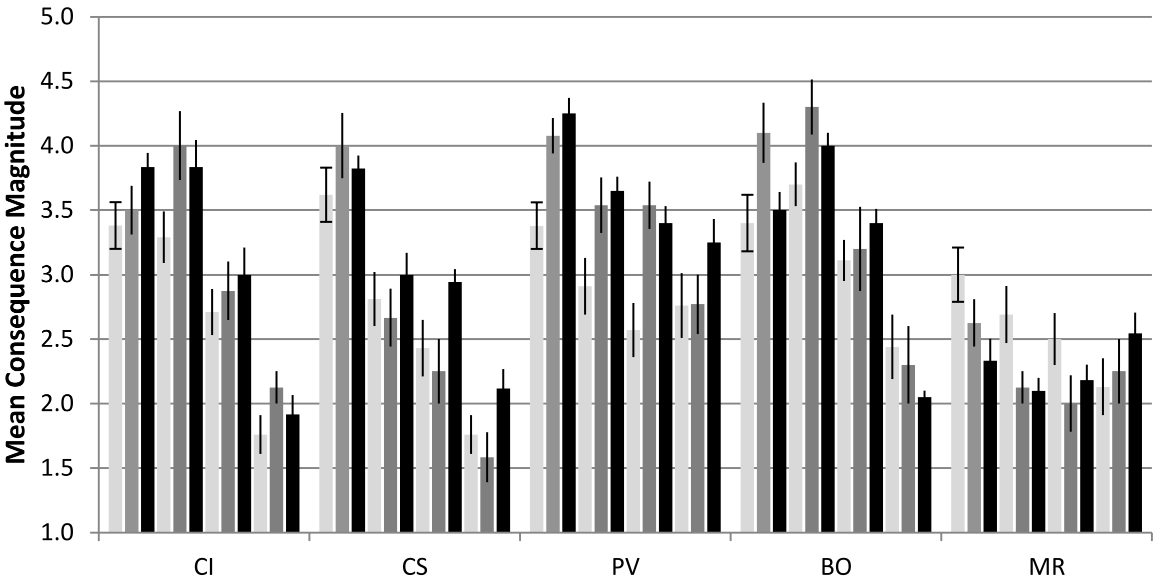


Figure 1 cont. Average consequence for each core value, for 'unknown species' for a) US/CA scientists; b) US/managers; c) AU scientists; and d) AU managers. UF=Unknown Fish; UG=Unknown Gastropod; UAsc=Unidentified Ascidian; UP=Unknown Parasite; UAlg=Unknown Algae. For each species, two bars at left=environmental; two bars second from left=economic; two bars second from right=social/cultural; and two bars at right=human health; grey=second assessment; and black=third assessment.

Appendix B5: Figure 2. Consequence estimates for known species.

a)



b)

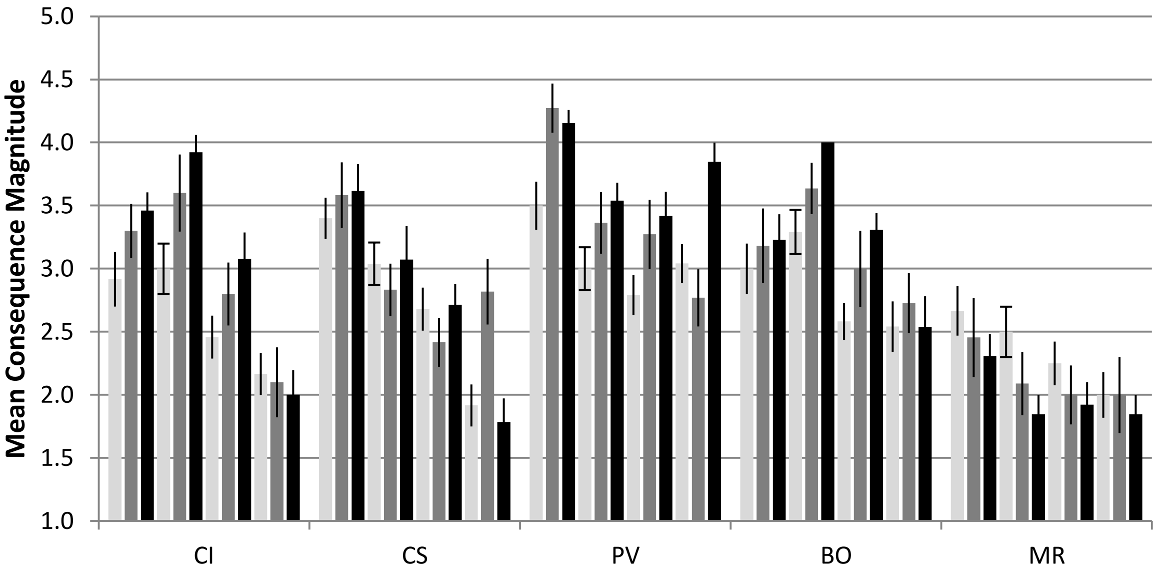
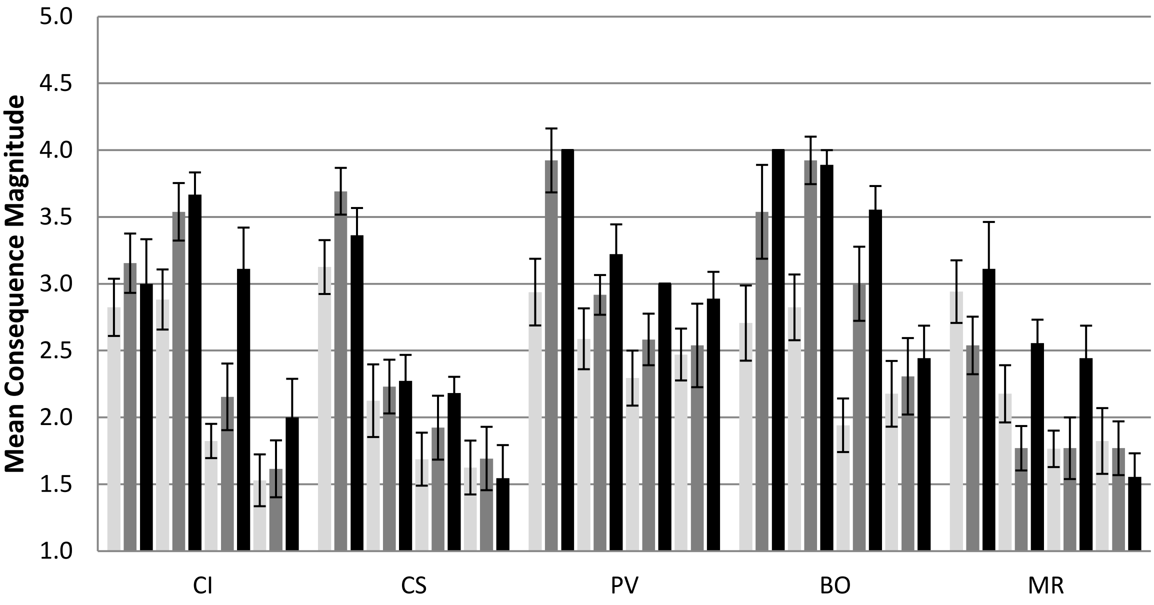


Figure 2. Average consequence for each core value, for 'known species' for a) US/CA scientists; b) US/CA managers; (overleaf) c) AU scientists; and d) AU managers. CI=*Ciona intestinalis*; CS=*Caulerpa scalpelliformis*; PV=*Pterois volitans*; BO=*Bonamia ostreae*; MR=*Maoricolpus roseus*. For each species, three bars at left=environmental; three bars second from left=economic; three bars second from right=social/cultural; and three bars at right=human health. Light grey=first assessment; medium grey=second assessment; and black=third assessment.

c)



d)

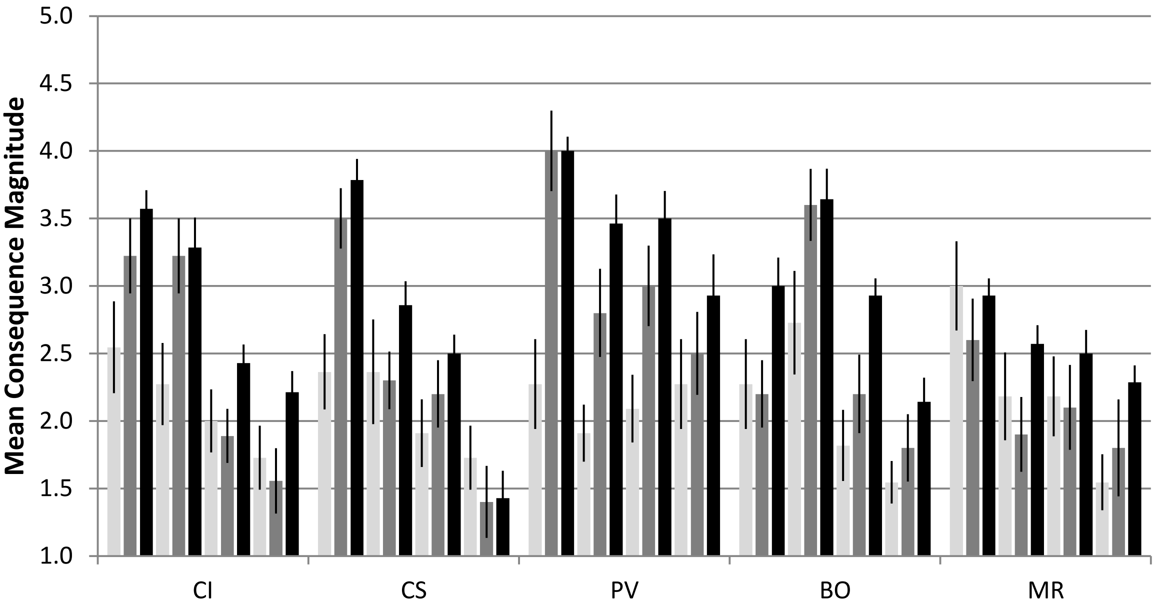


Figure 2 cont. Average consequence for each core value, for 'known species' for a) US/CA scientists; b) US/CA managers; c) AU scientists; and d) AU managers. CI=*Ciona intestinalis*; CS=*Caulerpa scalpelliformis*; PV=*Pterois volitans*; BO=*Bonamia ostreae*; MR=*Maoricolpus roseus*. For each species, three bars at left=environmental; three bars second from left=economic; three bars second from right=social/cultural; and three bars at right=human health. Light grey=first assessment; medium grey=second assessment; and black=third assessment.

Appendix C: Table 1. List of algal species included in review. Those with abbreviations had articles with analyses used in the review.

Species
Codium fragile (CF)
Caulerpa racemosa (CR)
Caulerpa taxifolia (CT)
Sargassum muticum (SM)
Undaria pinnatifida (UP)
Womersleyella setacea (WS)

Appendix C: Table 2. List of crustacean species included in review. Those with abbreviations had articles with analyses used in the review.

Species	Species cont.
Ampelisca abdita	*Gammarus tigrinus*
Anadara demiri	*Hemigrapsus penicillatus*
Alpheus inopinatus	*Hemigrapsus sanguineus* (HS)
Alpheus rapacida	*Homarus americanus*
Acartia tonsa	*Laticorophium baconi*
Briarosaccus callosus	*Ligia exotica*
Balanus eburneus (BE)	*Loxothylacus panopaei*
Balanus glandula	*Megabalanus coccopoma*
Balanus improvisus (BI)	*Pachygrapsus fakaravensis*
Callinectes bocourti	*Palaemon elegans*
Crangonyx floridanus	*Pseudodiaptomus inopinus*
Charybdis japonica	*Pseudodiaptomus forbesi*
Caprella mutica	*Paramysis lacustris*
Chthamalus proteus (CP)	*Pseudodiaptomus marinus*
Carcinoscorpius rotundicauda	*Pontogammarus robustoides*
Callinectes sapidus	*Rhithropanopeus harrisi*
Caprella scaura	*Sphaeroma annandalei*
Dikerogammarus haemobaphes	*Sinocalanus doerri*
Dikerogammarus villosus (DV)	*Solidobalanus fallax*
Echinogammarus berilloni	*Sylon hippolytes*
Echinogammarus ischnus	*Sphaeroma terebrans*
Eriocheir sinensis (ES)	*Tortanus dextrilobatus*
Gmelinoides fasciatus	

Appendix C: Table 3. List of algal literature. SM=*Sargassum muticum*, CF=*Codium fragile*, CR=*Caulerpa racemosa*, CT=*Caulerpa taxifolia* and UP=*Undaria pinnatifida*. S= significant analyses; NSP=nonsignificant analyses from which power can be calculated; and NSNP=nonsignificant analyses from which power cannot be calculated.

Citation	Species	S	NSP	NSNP
Airoldi, L. 2000. Effects of disturbance, life histories, and overgrowth on coexistence of algal crusts and turfs. Ecology **81**:798-814.	WS			X
Ambrose, R.F. and B.V. Nelson. 1982. Inhibition of giant kelp recruitment by an introduced brown alga. Botanica Marina **25**:265-268.	SM	X	X	
Britton-Simmons, K.H. 2004. Direct and indirect effects of the introduced alga *Sargassum muticum* on benthic, subtidal communities of Washington State, USA. Marine Ecology Progress Series **277**:61-78.	SM	X	X	
Bulleri, F., L. Airoldi, G. M. Branca, and M. Abbiati. 2006. Positive effects of the introduced green alga, *Codium fragile* ssp. *tomentosoides*, on recruitment and survival of mussels. Marine Biology **148**:1213-1220.	CF	X		
Ceccherelli, G. and D. Campo. 2002. Different effects of *Caulerpa racemosa* on two co-occurring seagrasses in the Mediterranean. Botanica Marina **45**:71-76.	CR	X		X
Ceccherelli, G. and F. Cinelli. 1997. Short-term effects of nutrient enrichment of the sediment and interactions between the seagrass *Cymodocea nodosa* and the introduced green alga *Caulerpa taxifolia* in a Mediterranean Bay. Journal of Experimental Marine Biology and Ecology **217**:165-177.	CT	X		
Ceccherelli, G. and N. Sechi. 2002. Nutrient availability in the sediment and the reciprocal effects between the native seagrass *Cymodocea nodosa* and the introduced rhizophytic alga *Caulerpa taxifolia*. Hydrobiologia **474**:57-66.	CT		X	
Ceccherelli, G., D. Campo, and L. Piazzi. 2001. Some ecological aspects of the introduced alga *Caulerpa racemosa* in the Mediterranean: way of dispersal and impact on native species. Biologia marina mediterranea **8**:94-99.	CR			X
Chavanich, S. and L. Harris. 2004. Impact of the non-native macroalga *Codium fragile* (Sur.) Hariot ssp. *tomentosoides* (van Goor) Silva on the native snail *Lacuna vincta* (Montagu, 1803) in the Gulf of Maine. Veliger **47**:85-90.	CF	X		
De Wreede, R. E. 1983. *Sargassum muticum* (Fucales, Phaeophyta): regrowth and interaction with *Rhodomela larix* (Ceramiales, Rhodophyta). Phycologia **22**:153-160.	SM	X		
Farrell, P. and R. L. Fletcher. 2006. An investigation of dispersal of the introduced brown alga *Undaria pinnatifida* (Harvey) Suringar and its competition with some species on the man-made structures of Torquay Marina (Devon, UK). Journal of Experimental Marine Biology and Ecology **334**:236-243.	UP			X

Appendix C: Table 3 cont.

Citation	Species	S	NSP	NSNP
Levin, P. S., J. A. Coyer, R. Petrik, and T. P. Good. 2002. Community-wide effects of nonindigenous species on temperate rocky reefs. Ecology **83**:3182-3193.			X	
Piazzi, L., D. Balata, G. Ceccherelli, and F. Cinelli. 2005. Interactive effect of sedimentation and *Caulerpa racemosa* var. *cylindracea* invasion on macroalgal assemblages in the Mediterranean Sea. Estuarine, Coastal and Shelf Science **64**:467-474.	CR	X		X
Piazzi, L. and G. Ceccherelli. 2006. Persistence of biological invasion effects: recovery of macroalgal assemblages after removal of *Caulerpa racemosa* var. *cylindracea*. Estuarine, Coastal and Shelf Science **68**:455-461.	CR	X		
Sánchez, Í. and C. Fernández. 2005. Impact of the invasive seaweed *Sargassum muticum* (phaeophyta) on an intertidal macroalgal assemblage. Journal of Phycology **41**:923-930.	SM	X	X	X
Scheibling, R. E. and P. Gagnon. 2006. Competitive interactions between the invasive green alga *Codium fragile* ssp. *tomentosoides* and native canopy-forming seaweeds in Nova Scotia (Canada). Marine Ecology Progress Series **325**:1-14.	CF	X	X	
Schmidt, A. L. and R. E. Scheibling. 2007. Effects of native and invasive macroalgal canopies on composition and abundance of mobile benthic macrofauna and turf-forming algae. Journal of Experimental Marine Biology and Ecology **341**:110-130.	CF	X	X	
Valentine, J. P. and C. R. Johnson. 2005. Persistence of the exotic kelp *Undaria pinnatifida* does not depend on sea urchin grazing. Marine Ecology Progress Series **285**:43-55.	UP	X		X
Viejo, R. M. 1997. The effects of colonization by *Sargassum muticum* on tidepool macroalgal assemblages. Journal of the Marine Biological Association of the United Kingdom **77**:325-340.	SM	X	X	X

Appendix C: Table 4. List of crustacean literature. BI=*Balanus improvisus*, BE=*Balanus eburneus*, HS=*Hemigrapsus sanguineus*, DV= *Dikerogammarus villosus,* ES= *Eriocheir sinensis*, CT= *Chthamalus proteus.* S= significant analyses; NSP=nonsignificant analyses from which power can be calculated; and NSNP=nonsignificant analyses from which power cannot be calculated.

Citation	Species	S	NSP	NSNP
Barnes, B. B., M. W. Luckenbach, and P. R. Kingsley-Smith. 2010. Oyster reef community interactions: the effect of resident fauna on oyster (*Crassostrea* spp.) larval recruitment. Journal of Experimental Marine Biology and Ecology **391**:169-177.	BI	X		
Boudreaux, M., L. Walters, and D. Rittschof. 2009. Interactions between native barnacles, non-native barnacles, and the eastern oyster *Crassostrea virginica*. Bulletin of Marine Science **84**:43-57.	BE	X		
Brousseau, D. J. and R. Goldberg. 2007. Effect of predation by the invasive crab *Hemigrapsus s*anguineus on recruiting barnacles *Semibalanus balanoides* in western Long Island Sound, USA. Marine Ecology Progress Series **339**:221-228.	HS	X	X	
Dürr, S. and M. Wahl. 2004. Isolated and combined impacts of blue mussels (*Mytilus edulis*) and barnacles (*Balanus improvisus*) on structure and diversity of a fouling community. Journal of Experimental Marine Biology and Ecology **306**:181-195.	BI			X
Griffen, B. 2006. Detecting emergent effects of multiple predator species. Oecologia **148**:702-709.	HS			X
Kotta, J. et al. 2006. Ecological consequences of biological invasions: three invertebrate case studies in the north-eastern Baltic Sea. Helgoland Marine Research **60**:106-112.	BI	X		
Lohrer, A. M. and R. B. Whitlatch. 2002. Relative impacts of two exotic brachyuran species on blue mussel populations in Long Island Sound. Marine Ecology Progress Series **227**:135-144.	HS	X		
Platvoet, D., J. T. A. Dick, N. Konijnendijk, and G. van der Velde. 2006. Feeding on micro-algae in the invasive Ponto-Caspian amphipod *Dikerogammarus villosus* (Sowinsky, 1894). Aquatic Ecology **40**:237-245.	DV	X		
Rudnick, D. and V. Resh. 2005. Stable isotopes, mesocosms and gut content analysis demonstrate trophic differences in two invasive decapod crustacea. Freshwater Biology **50**: 1323-1336.	ES	X	X	
Tyrrell, M. C., P. A. Guarino, and L. G. Harris. 2006. Predatory impacts of two introduced crab species: inferences from microcosms. Northeastern Naturalist **13**:375-390.	HS	X	X	
Young, C. M. and J. L. Cameron. 1989. Differential predation by barnacles upon larvae of two bryozoans: spatial effects at small scales. Journal of Experimental Marine Biology and Ecology **128**: 283-294.	BE	X		
Zabin, C. J. 2005. Community ecology of the invasive intertidal barnacle *Chthamalus proteus* in hawai'i. Department of Zoology, University of Hawai'i.	CP	X		

Appendix C: Table 4 cont.

Citation	Species	S	NSP	NSNP
Zabin, C. J. and A. Altieri. 2007. A Hawaiian limpet facilitates recruitment of a competitively dominant invasive barnacle. Marine Ecology Progress Series **337**:175-185.	CP	X		

Appendix C: Table 5. Significant results for nonindigenous algal abundance impact studies. SM=*Sargassum muticum*, CF=*Codium fragile*, CR=*Caulerpa racemosa*, CT=*Caulerpa taxifolia* and UP=*Undaria pinnatifida.* PC=percent cover.

Citation	Species	Location	Experimental methods	Results
Ambrose and Nelson (1982)	*Sargassum muticum*	Santa Catalina, California	Removal of SM to test effects on density for *Macrocystis pyrifera*	Greater density in removal treatments for *M. pyrifera* [plants/m2±SD], June 1979, site1: removal=5±5.91, control=.4±.63; site2a: removal=1.9±1.38, control=.2±.37; September 1979, site1: removal=1.9±2.06, control=.1±.31; site2a: removal=.4±.6, control=0±0; site2b: removal=.1±.31, control=0±0
Britton-Simmons (2004)	*Sargassum muticum*	San Juan Islands, Washington State	Removal of SM to rest effects on PC of native canopy algae, understory algae and urchin *Strongylocentrotus droebachiensis*	Less abundant canopy algae (F=22.86, p=0.000) and understory algae (F=8.81, 0.009) in removal treatments; more abundant *S. droebachiensis* (t=-6.34, p<0.001) in removal treatments.
Bulleri et al (2006)	*Codium fragile*	Italy, Adriatic Sea	Removal of CF to test effects on density of *Mytilus galloprovincialis*	Lesser density in removal treatments than on bare rock (F=6.20)
Ceccherelli and Campo (2002)	*Caulerpa racemosa*	Italy, Mediterranean Sea	Removal of CR to test effects on shoot density of *Zostera noltii*	Lesser (though variable) shoot density of *Z. noltii* in CR removal treatment (F=1881.74)
Ceccherelli and Cinelli (1997)	*Caulerpa taxifolia*	Galenzana Bay, Italy	Removal of CT to test effects on shoot density of *Cymodocea nodosa*	Greater shoot density of *C. nodosa* in CT removal treatment (F=8.506)
Chavanich and Harris (2004)	*Codium fragile*	Maine and New Hampshire	Reciprocal transplant of *Codium* and *Laminaria* to test effects on density of *Lacuna vincta*	Greater density of *L. Vincta* in *Laminaria* treatments (*Laminaria*: 17.21 snails/g algae; *Codium*: 2.7 snails/g algae
De Wreede (1983)	*Sargassum muticum*	British Columbia, Canada	Removal of SM to test for effects on PC of *Rhodomela larix* and articulated corraline algae	Greater PC of *R. larix* in removal treatments; lesser PC of articulate corraline algae in removal treatments

Appendix C: Table 5 cont.

Citation	Species	Location	Experimental methods	Results
Levin et al (2002)	*Codium fragile*	Isles of Shoals, New Hampshire	Removal of CF to test for effects on PC and density of native kelps and mobile animal species	Greater PC of kelp in removal treatments (F= 4.496); greater density of cunner *Tautogolabrus adspersus* in removal treatments (6-fold)
Piazzi and Ceccherelli (2006)	*Caulerpa racemosa*	Italy, Mediterranean Sea	Removal of CR to test effects on PC for encrusting, turf and erect algae	Greater PC in removal treatments for encrusting (F=723.89); turf (F=13.57); and erect (F=14.44) algae
Piazzi et al (2005)	*Caulerpa racemosa*	Italy, Mediterranean Sea	Addition of CR to test effects on PC of turf, erect and prostrate algae, as well as several specific species within these	Greater PC in non-CR treatments of erect and prostrate algae (F=119.5, 101.2) as well as for *Halimeda tuna* (F=26.88)
Scheibling and Gagnon (2006)	*Codium fragile*	Nova Scotia, Canada	Removals of CM in press (monthly) and pulse (annual) treatments to test effects on PC of kelps (*Laminaria longicruris* and *Laminaria digitata*) and algae (*Desmarestia viridis* and *D. aculeata*) and kelp *Saccorhiza dermatodea*.	Greater percent cover of *Laminaria* and *Desmarestia* after first pulse removal (F=14.5, 22.9); after half of press and second pulse removals (4.8, 14.8 - greater than control but no difference between removal treatments); and *Laminaria* at end of experiment (F=58.2, with press greater than pulse - *Desmarestia* not mentioned)
Schmidt and Scheibling (2007)	*Codium fragile*	Nova Scotia, Canada	Repeated measures removal of CF from breakwaters dominated by CF to test effects on PC of coarsely branched algae, jointed calcerous algae, sheet forming algae, filamentous algae and benthic macrofauna	Lesser density of benthic macrofauna in CF removal treatment: *P. gunnellus* [mean±SE (individuals m−2]; Canopy Intact (CI): 0.17±0.13, Canopy Removed (CR): 0.03±0.06]; *Asterias* spp. (CI: 0.75±0.35, CR: 0.2±0.16); *C. irroratus* (CI: 0.59±0.36, CR: 0.34±0.28); and *Pagurus* spp. (CI: 4.55±1.2, CR: 2.38±0.8)
Valentine and Johnson (2005)	*Undaria pinnatifida*	Tasmania, Australia	Removals of UP to test effects on PC of total native algae, red algae, native canopy-forming algae, green algae and brown turf algae	Greater percent cover after removal for green and brown turf algae (F= 4.20, 6.88 respectively) in first sample date only

Appendix C: Table 5 cont.

Citation	Species	Location	Experimental methods	Results
Viejo (1997)	*Sargassum muticum*	Spain, East Atlantic	Removals of SM to test effects on PC of foliose algae, filamentous algae, coarsely branched macrophytes, leathery macrophytes, articulated calcerous and articulate crustose algae.	Greater percent cover of total native and leathery algae greater in non-SM site 3 (F=6.23, 6.29 respectively; no ES)

Appendix C: Table 6. Significant results for nonindigenous crustacean abundance impact studies. BI=*Balanus improvisus*, BE=*Balanus* eburneus, HS=*Hemigrapsus sanguineus*, DV= *Dikerogammarus villosus*, ES= *Eriocheir sinensis*, CT= *Chthamalus proteus*. PC=percent cover.

Citation	Species	Location	Experimental methods	Results
Barnes et al (2010)	*Balanus improvisus*	Chesapeake Bay, Maryland	Varying PC of BI (no, low, medium and high) to test effects on density (settlement) of *Crassostrea virginica* and *Crassostrea ariakensis*.	Difference in density (settlement), with lower density on control (absence of BI) treatments than for low, medium or high treatments for both *C. ariakensis* (F=5.82) and *C. virginica* (F=11.00).
Boudreaux et al (2009)	*Balanus eburneus*	Florida	Varying densities of BE (control/0%, low/25%, medium/25-50% and high/>50%) to test effects on density (settlement) of *Crassostrea virginica*.	Difference in density (settlement) (F=3.545), with greatest density in absence of barnacles, followed by low barnacle levels
Brousseau and Goldberg (2007)	*Hemigrapsus sanguineus*	Long Island Sound, Connecticut	Used presence/absence treatments: no crabs, low density (15 crabs), medium density (45 crabs) and high density (90 crabs) to determine effect on density of *Semibalanus balanoides*	Greater densities of *S. balanoides* in no and low crab treatments (F=2.67) in middle of experiment, though no difference by end of experiment.
Kotta et al (2006)	*Balanus improvisus*	Baltic Sea	Varying PC of BI (0, 10, 20, 40, 70, 80 and 100%) to test effects on density of *Enteromorpha intestinalis*	Greater density of *E. intestinalis* was found with increasing PC of BI (r^2=0.91)
Lohrer and Whitlatch (2002)	*Hemigrapsus sanguineus*	New Hampshire	Used three presence/absence experiments to determine effect of HS on density of *Mytilus edulis*. The second and third were one year later, and the third used greater densities of smaller crabs.	Lower densities of *M. edulis* in presence treatments for both experiments (~25-60% decrease).
Platvoet et al (2006)	*Dikerogammarus villosus*	The Netherlands	Presence/absence treatments to tested effect of 3 groups of DV (adult male, adult female and juvenile) and control (no DV) on density of *Monoraphidium griffithii*.	Greater concentration in control treatment (F=10.3), with no differences between DV-present groups.

Appendix C: Table 6 cont.

Citation	Species	Location	Experimental methods	Results
Rudnick and Resh (2005)	*Eriocheir sinensis*	San Francisco, California	Presence/absence treatments to test effect on oligochaetes, *Trichoptera* (caddisfly *Gumaga nigricula*), *Ephemeroptera* (Trichorythidae and Leptophlebiidae), large and small *Corbicula fluminae*, algae and macrophytes.	Lower density of the caddisfly *Gumaga* (by 90%) and *C. fluminae* (by 50%). Greater abundance of oligochaetes (>150%). Also, greater biomass of algae (F = 19.7) and lower biomass of detritus (F = 40.1 for *Salix*, 33.8 for *Platanus*).
Tyrrell et al (2006)	*Hemigrapsus sanguineus*	New Hampshire	Used varying densities in lab (control and 2 crabs/cage) and short and long field (control and 5 crabs/cage) to test effect on *Semibalanus balanoides*, *Spirorbis* sp and ephemeral, crustose and fucoid algae.	Great densities of *S. balanoides* in control treatments for lab [PC control, PC present] (5.56, -85.25) and field (short: -2.73, -35.45; long: -6.70, -83.84) experiments. Greater densities of *Spirorbis* sp (0, -83.36) but lower crustose (-3.00, 10.74) algae PC in control lab experiments
Young and Cameron (1989)	*Balanus eburneus*	Florida	Presence of live/dead BE to test effects on density (settlement) of bryozoans *Bugula neritina* and *B. stolonifera*	Presence of live barnacles lead to greater settlement than dead barnacles (F=4.81)
Zabin (2005)	*Chthamalus proteus*	Hawaii	Removal treatments to determine effect on PC and on density (settlement) of *Balanus reticulatus*	Greater PC (F=96.79) and density (F=18.72) of *B. reticulatus* in CP removal treatments
Zabin and Altieri (2007)	*Chthamalus proteus*	Hawaii	Removal treatments (press, pulse and control) to test effect on *Siphonaria normalis*	Greater density in CP removal treatments (F=27.59), with greatest density in press removal, followed by pulse removal then no removal.

Appendix C: Table 7. Power calculations for non-significant nonindigenous algae abundance impact studies. [1] indicates power analysis via test (otherwise via ANOVA). SM=*Sargassum muticum*, CF=*Codium fragile*, CR=*Caulerpa racemosa*, CT=*Caulerpa taxifolia* and UP=*Undaria pinnatifida.* PC=percent cover.

Citation	Species	Location	Experimental methods	Results
Ambrose and Nelson (1982)	*Sargassum muticum*	Santa Catalina, California	Removal of SM to test effects on density of *Macrocystis pyrifera*	No effect[1] on density of *M. pyrifera* for site 2b in June 1979 (ES=0.34, power=0.17) or September 1979 (ES=0.46, power-0.28)
Britton-Simmons (2004)	*Sargassum muticum*	San Juan Islands, Washington State	Removal of SM to rest effects on PC of bare rock, crustose corraline algae and turf-forming algae	No effect on PC of bare rock (author-calculated power=0.05), crustose corraline algae (author calculated power=0.2), or turf-forming algae (author-calculated power=0.4).
Ceccherelli and Sechi (2002)	*Caulerpa taxifolia*	Italy (Galenzana Bay)	Removal of CT to test effects on shoot density of *Cymodocea nodosa*	No effect on density of *C. nodosa* (ES=3.62, power=1).
Levin et al (2002)	*Codium fragile*	Gulf of Maine	Removal of CF to test for effects on PC and density of native kelps and mobile animal species	No effect on density of *Cancer irroratus* (ES=0, power=0.05), *Carcinus maenas* (ES=0.043, power=0.050) or *Homarus americanus* (ES=0.052, power-0.053)
Sanchez and Fernandez (2005)	*Sargassum muticum*	Spain, East Atlantic	Removal treatments of SM to test for successional effects on PC of dominant macroalgal species. Both treatments started as bare rock, with removals of SM as one treatment and non-removal as control.	No effect on PC of *Bifurcaria bifurcata* (ES=0.35, power=0.32) or *Gelidium spinosum* (ES=0.24, power=0.18)
Scheibling and Gagnon (2006)	*Codium fragile*	Nova Scotia, Canada	Removals of CM in press (monthly) and pulse (annual) treatments to test effects on PC of kelps (*Laminaria longicruris* and *Laminaria digitata*) and algae (*Desmarestia viridis* and *D. Aculeata*) and kelp *Saccorhiza dermatodea*.	No effect on density of *Saccorhiza dermatodea* (ES=0.42, power=0.23)

Appendix C: Table 7 cont

Citation	Species	Location	Experimental methods	Results
Schmidt and Scheibling (2007)	*Codium fragile*	Nova Scotia, Canada	Repeated measures removals of CF from breakwaters dominated by CF to test effects on PC of coarsely branched algae, jointed calcerous algae, sheet forming algae, filamentous algae and benthic macrofauna	No effect on PC of *Myoxocephalus scorpius* (ES=0.16, power=0.13).
Viejo (1997)	*Sargassum muticum*	Spain, East Atlantic	Removals of SM to test effects on PC of foliose algae, filamentous algae, coarsely branched macrophytes, leathery macrophytes, articulated calcerous and articulate crustose algae.	No effect[1] on any group, specifically crustose algae at site 1 (ES=0.43, power=0.08), 2 (ES=0.37, power=0.08), or 3 (ES=0.10, power=0.05); or *Bifurcaria bifurcata* site 1 (ES=0.47, power=0.09) or 2 (0.50, power=0.09)

Appendix C: Table 8. Power calculations for non-significant nonindigenous crustacean abundance impact studies. BI=*Balanus improvisus*, BE=*Balanus eburneus*, HS=*Hemigrapsus sanguineus*, DV= *Dikerogammarus villosus,* ES= *Eriocheir sinensis*, CT= *Chthamalus proteus.* PC=percent cover.

Citation	Species	Location	Experimental methods	Results
Brousseau and Goldberg (2007)	*Hemigrapsus sanguineus*	Long Island Sound, Connecticut	Used presence/absence treatments in 2 experiments: (1): exclosure, enclosure (15 crabs/0.25m^2), partial open, and open cages in 2003 and 2004; and (2): no crabs, low density (15 crabs/0.25m^2), medium density (45 crabs/0.25m^2) and high density (90 crabs/0.25m^2) cages in 2005 to determine effect on density of *Semibalanus balanoides* and *Ulva* spp. (in experiment 2)	No significant effect of HS on *S. balanoides* in 2003 experiment 1 (ES=0.65, power=0.43) or 2005 experiment 2 (ES=0.89, power=0.72). Despite strong trend of decreasing PC of *Ulva* spp with higher crab densities, no significant effect of HS (ES=0.70, power=0.49).
Rudnick and Resh (2005)	*Eriocheir sinensis*	San Francisco	Presence/absence treatments to test effect on oligochaetes, Trichoptera (caddisfly *Gumaga nigricula*), Ephemeroptera (Trichorythidae and Leptophlebiidae), large and small *Corbicula fluminae*, algae and macrophytes.	No significant effect on change in PC of macrophytes (*Ludwigia*) (ES=0.29, power=0.17
Tyrrell et al (2006)	*Hemigrapsus sanguineus*	New Hampshire	Used varying densities in lab (control and 2 crabs/cage) and short and long field (control and 5 crabs/cage) to test effect on change in percent cover for *Semibalanus balanoides*, *Spirorbis* sp and ephemeral, crustose and fucoid algae, and *Mastocarpus/Chondrus* spp.	No significant effect of HS on fucoid algae (ES=0.30, power=0.37) or *Mastocarpus/Chondrus* (ES=0.13, power=0.10) in lab; on *Spirorbis* sp (ES=0.47, power=0.57), *Mytilus edulis* (ES=0.17, power=0.11), ephemeral (ES=0.17, power=0.12), crustose (ES=0.23, power=0.17) or fucoid (ES=0.17, power=0.11) algae or *Mastocarpus/Chondrus* (ES=0.13, power=0.09) in 2-day field; or on *Spirorbis* (ES=0.73, power=0.67), ephemeral (ES=0.75, power=0.70) or crustose (ES=0.39, power=0.23) algae, or *Mastocarpus/Chondrus* (ES=0.64, power=0.54) in 14-day field experiment.

Appendix C: Table 9. Non-significant results for nonindigenous algal abundance impact studies. SM=*Sargassum muticum*, CF=*Codium fragile*, CR=*Caulerpa racemosa*, CT=*Caulerpa taxifolia* and UP=*Undaria pinnatifida*. PC=percent cover.

Citation	Species	Location	Experimental methods	Results
Airoldi (2000)	*Womersleyella setacea (turf)*	Ligurian Sea, Calafuria, Italy	Removal of WS (initial removal of turf; repeated removal of turf; abrasion of turf; control) to test effects on crust algae	No effect on PC of crust algae ($F=2.08$, $p>0.05$).
Ceccherelli and Campo (2002)	*Caulerpa racemosa*	Italy, Mediterranean Sea	Removal of CR to test effects on shoot density of *Cymodocea nodosa*	No effect($F=5.49$, $p=0.1438$)
Ceccherelli et al (2001)	*Caulerpa racemosa*	Mediterranean (Italy)	Removals of CR to test effects on shoot density of *Cymodocea nodosa*	No effect ($p=0.1438$)
Farrell and Fletcher (2006)	*Undaria pinnatifida*	Devon, UK	Removals of UP to test effects on PC of red and green algae, and density of *Styela clava*	No effect on any group ("NS")
Piazzi et al (2005)	*Caulerpa racemosa*	Italy, Mediterranean Sea	Addition of CR to test effects on PC of turf, erect and prostrate algae, as well as several specific species within these	No effect on PC of turf ($F=1.26$) or density of *Flabellia petiolata* (10.25), *Laurencia obtusa* (8.04) and *Peyssonnelia rubra* (4.93) ($p>0.05$).
Sanchez and Fernandez (2005)	*Sargassum muticum*	Spain, East Atlantic	Removal of SM to test for effects on PC of dominant macroalgal species	No effect on PC of *Bifurcaria bifurcata* (λ-1.85, p=.203); *Gelidium spinosum* ($\lambda=2.20$, $p=0.169$); or rest of species ($\lambda=0.51$, $p=0.488$).
Valentine and Johnson	*Undaria pinnatifida*	Tasmania, Australia	Removals of UP at two difference dates to test effects on PC of total native algae, red algae, native canopy-forming algae, green algae and brown turf algae	No effect on PC at either date for total native algae [2000:F,p; 2001:F,p] (1.21, 0.157; 2.5, 0.125), red algae (1.07; 0.31; 3.05, 0.092) or native canopy-forming algae (1.63, 0.213; 0.32, 0.577). No effect on PC in 2001 for green algae [F,p] (0.1, 0.76) or brown turf algae (0, 0.963)

Appendix C: Table 10. Non-significant results for nonindigenous crustacean abundance impact studies. PC=percent cover.

Citation	Species	Location	Experimental methods	Results
Durr and Wahl (2004)	*Balanus improvisus*	Baltic Sea	Removal treatments to determine effect on PC (small specimens) or density (large specimens) of *Zoothamnium* spp, *Laomedea flexuosa*, *Ceramium strictum*, *Folliculina* spp, *Mytilus edulis*, *Membranipora crustulenta*, *Polydora* spp, *Corophuim* spp, and *Clava multicornis*	No effect on any group (F, p not given)
Griffen (2006)	*Hemigrapsus sanguineus*	New Hampshire	Presence/absence treatments to test effect on single and two different densities (low and high) of *Mytilus edulis*	No effect on either single (F=0.005, p=0.946) or two different densities of *M. edulis* (F=0.036, p=0.850).

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